ICES WGMME REPORT 2014

ICES ADVISORY COMMITTEE

ICES CM 2014/ACOM:27

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

10–13 March 2014

Woods Hole, Massachusetts, USA

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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Recommended format for purposes of citation:

ICES. 2014. Report of the Working Group on Marine Mammal Ecology (WGMME), 10–13 March 2014, Woods Hole, Massachusetts, USA. ICES CM 2014/ACOM:27. 228 pp.

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1 Introduction

The Working Group on Marine Mammal Ecology (WGMME) met at Woods Hole Oceanographic Institution (WHOI), Quissett Campus, Massachusetts, USA from 10 March to 14 February 2014. A satellite meeting was held in Oban, Scotland from 11 March to 13 March 2014 specifically to consider the Term of Reference on monitoring for renewable installations (ToR f). During plenary, the two meetings were linked through video skype. The list of participants and contact details for both meetings are given in Annex 1.

On behalf of the working group, the chair would like to thank the efforts and support provided by both Gordon Waring (NOAA) and Michael Moore (WHOI). In particular, their help in organising the meeting and ensuring that everything ran smoothly, including links to the satellite meeting in Oban. Thanks are also due to Barbara Newell for helping to organise the accommodation requirements for the meeting. The chair would also like to thank Fred Serchuk (ICES Vice President and US Delegate) for presenting the ICES Strategic Plan to the working group. Thanks also to Michael Moore and Michael Simpkins for introducing the work undertaken at WHOI and by the NOAA Protected Species Branch. Thanks are also due to Chris Orphanides, Marjorie Lyssikatos, Allison Henry, Tim Cole, Richard Pace, Dani Cholewiak and Beth Josephson for taking time to present their research to the working group.

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. These included Callan Duck, Sophie Brasseur, Cecile Vincent, Anita Gilles, Signe Sveegaard, Norbert Dankers, Vincent Ridoux, Jorge Manuel Bastos Santos, Michelle Cronin, Mark Jessops, José Vingada, Marina Sequeira, Tero Härkönen, Oliver Ó Cadhla, Ferdia Marnell, Jérôme Spitz, Ursula Seibert, Simon Berrow and Robin Law.

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 Terms of Reference 2014

The following Terms of Reference and the work schedule were adopted on 10 March 2014.

a) Review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals; specifically. This will contribute to the work required for the MoU between the European Commission and ICES to "*provide new information regarding the impact of fisheries on other components of the ecosystem including small cetaceans and other marine mammals*…" and to aid "*scientific and technical developments in the support of the Marine Strategy Framework Directive, such as by designing marine monitoring and assessment programmes, identifying research needs and methodologies advice*". OSPAR is also seeking advice from ICES in relation to the development of indicators and targets for determining Good Environmental Status (GES) under MSFD to which this will contribute;

b) Provide information on abundance, distribution, population structure and incidental capture of marine mammals in the western North Atlantic (North Atlantic right whale, harbour porpoise and white-sided dolphin);

c) To review the further development of the Bycatch Limit Algorithm framework for determining safe bycatch limits. This work should include harbour porpoise, short-beaked common dolphin and consideration of additional species for which bycatch estimates have been made or suggested as potential MSFD indicators (e.g. bottlenose dolphin, striped dolphin, harbour seal and grey seal). This should include a comparison with approaches used to assess bycatch in USA;

d) Assess the Joint Cetacean Protocol outputs with a view to their contribution to international transboundary reporting requirements (e.g. for Article 17 of the Habitats Directive) and the operationalization of MSFD indicators, targets and appropriate baselines. Consideration should also be given to other approaches, such as those of the Atlantic marine Assessment programme (AMAPPS) which coordinates data collection and analysis for marine mammals and reptiles for population assessments;

e) Update on development of database for seals and status of intersessional work, contribution to the operationalization of MSFD indicators, targets and appropriate baselines. Consideration should also be given to other approaches, such as those of the Atlantic Marine Assessment programme (AMAPPS);

f) Outline and review approaches to marine mammal survey design used during pre- and post-consenting monitoring in the offshore marine renewables (wind, wave, tide) industry, and provide recommendations for best practice.

g) Special request:

Interactions between wild and captive fish stocks (OSPAR 4/2014)

a) Recalling the conclusion of the QSR 2010 that mariculture is a growing activity in the OSPAR maritime area, EIHA 2012 considered the potential for increasing environmental pressure relating to the growth of this industry. As yet this is not an established work stream within EIHA, and Contracting Parties have requested that more information be brought forwards on this issue. This was reiterated by EIHA 2013.

b) Mariculture has a number of associated environmental pressures such as the introduction of non-indigenous species, which can have ecological and genetic impacts on marine environment and especially on wild fish stocks; in addition, pressures from mariculture might include:
    i) introduction of antibiotics and other pharmaceuticals;
    ii) transfer of disease and parasite interactions;
    iii) release of nutrients and organic matters;
    iv) introgression of foreign genes, from both hatchery-reared fish and genetically modified fish and invertebrates, in wild populations;
    v) effects on small cetaceans, such as the bottlenose dolphin, due to their interaction with aquaculture cages
c) EIHA proposes that OSPAR requests ICES to provide:
    i) an update on the available knowledge of these issues;
    ii) concrete examples of management solutions to mitigate these pressures on the marine environment;
    iii) advice on which pressures have sufficient documentation regarding their impacts to implement relevant monitoring and suggest a way forward to manage these pressures.
d) It may be appropriate to explore cooperation with other competent authorities working in this field, such as the European Food Safety Authority with respect to disease transfer or parasites, or the North Atlantic Salmon Conservation Organisation (NASCO), in particular with respect to existing cooperation between NASCO and ICES on issues pertaining to pressures from mariculture.

WGMME is requested in particular to address point bv. Also WGAQUA, WGPDMO and WGAGFM will address this request.

h) Special request: Marine mammals (OSPAR 6/2014)
    - Advise on appropriate management units (MUs) for grey and harbour seals in the OSPAR Maritime area;
    - Provide technical and scientific advice on options for ways of setting targets for the OSPAR common MSFD Indicators for marine mammals and where possible, provide examples of the application of these options. The advice should consider the suitability of various options for relevant marine mammal species/ MUs/ indicators. In considering target setting options, also consider the consequences that this may have for the monitoring programme (including spatial and temporal implications). Consideration should be given to precision in target setting and monitoring. (Note that ICES are not asked to take any societal/ policy choices, but if necessary should identify the need for such choices and their potential implications);
    - Provide an overview of existing monitoring per OSPAR common MSFD indicator and marine mammal species, including the description of current monitoring frequency (and whether this is likely to be sufficient to meet the assessment requirement);

- Provide an overview of possible future monitoring requirements and methodology per OSPAR common MSFD indicator and marine mammal species.
- The request is to cover OSPAR regions II, III and IV.
- The existing indicator technical specifications developed by COBAM should form the basis of this work.

WGMME will report to the attention of the Advisory Committee (ACOM) by 4 April 2014.

Supporting Information

Priority	The current activities of this Group will lead ICES into issues related to the ecosystem affects of fisheries, especially with regard to the application of the Precautionary Approach. Consequently, these activities are considered to have a very high priority.
Scientific justification	Term of Reference a) This will contribute to the work required for the MoU between the European Commission and ICES to "*provide new information regarding the impact of fisheries on other components of the ecosystem including small cetaceans and other marine mammals…*" and to aid "*scientific and technical developments in the support of the Marine Strategy Framework Directive, such as by designing marine monitoring and assessment programmes, identifying research needs and methodologies advice*". OSPAR is also seeking advice from ICES in relation to the development of indicators and targets for determining Good Environmental Status (GES) under MSFD to which this will contribute. Term of Reference b) This will be a useful comparison with work off Europe and is appropriate considering the location of the meeting. Term of Reference c) This should include a comparison with approaches used to assess bycatch in USA. Term of Reference d) This work may aid relevant EU Member States in carrying out the requirements of MSFD, particularly, regional scale (i.e. transboundary) reporting. Term of Reference e) The seal database should be useful for future advice and in helping Member States meet some requirements of MSFD. WGMME are invited to discuss data management with the ICES DataCentre. Term of Reference f) There are many studies of the effects of renewable energy installations, both before and after construction. ICES recommendations could help ensure that these studies are effective and allow resources to be used effectively. Term of Reference g) This is part of a special request from OSPAR that, with input from other expert groups, will be used in ICES advice. Term of Reference h) This is part of a special request from OSPAR that, with input from other expert groups, will be used in ICES advice.
Resource requirements	Two rooms in the host institute; wifi or web access also essential.
Participants	The Group is normally attended by some 10–15 members.
Secretariat facilities	None.
Financial	No financial implications.
Linkages to ACOM and its expert groups	ACOM is the parent committee of WGMME. There are linkages to the work of WGECO and WGBYC.
Linkages to SCICOM and its expert groups	There are links to other groups looking at the effects of aquaculture.
Linkages to other organisations	

3 ToR A Review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals; specifically. This will contribute to the work required for the MoU between the European Commission and ICES to “provide new information regarding the impact of fisheries on other components of the ecosystem including small cetaceans and other marine mammals…” and to aid “scientific and technical developments in the support of the Marine Strategy Framework Directive, such as by designing marine monitoring and assessment programmes, identifying research needs and methodologies advice”. OSPAR is also seeking advice from ICES in relation to the development of indicators and targets for determining Good Environmental Status (GES) under MSFD to which this will contribute

In the previous years WGMME has discussed the development of management units (MUs) for harbour porpoise, common dolphin, bottlenose dolphin, white-beaked dolphin, white-sided dolphin and minke whale. Based on the report of the joint ASCOBANS-Helcom small cetacean population structure workshop (Evans and Teilmann, 2009) and work of the IWC Scientific Committee, WGMME (2012) made recommendations for MUs for these species but did not explicitly specify their boundaries. WGMME (2013) specified proposed boundaries of the MUs. New information received this year allowed this work to be taken further. The results are presented in Section 10 (ToR H).

Parallel with the development of management units for cetaceans WGMME (2012 and 2013) also considered management units for harbour and grey seals under the ToR to develop biodiversity indicators to inform the ongoing work of OSPAR-COBAM and MSFD. New information received this year allowed this work to be taken further. The results are presented in Section 10 (ToR H) as part of the OSPAR special request on the Marine Strategy Framework Directive.

3.1 New survey and abundance information

3.1.1 Distribution and abundance of harbour porpoise in the Kattegat, Belt Seas and western Baltic

WGMME (2013) reported on a shipboard survey conducted between 2 and 21 July 2012 at the same time of the year and along the same transects as SCANS-II in the Kattegat, Belt Seas and western Baltic (waters of Denmark, Sweden and Germany) covering the so-called gap-area and Methods and equipment that were identical with those in SCANS-II were used, including beyond (Figure 3.1) double platform data collection. Due to the differences in the boundaries of the survey areas, the results are not directly comparable to the previous survey in 2005 (SCANS-II, Hammond *et al.*, 2013), and 1994 (SCANS, Hammond *et al.*, 2002). The results of this survey have recently been published and discussed in Viquerat *et al.*, (2014).

Weather conditions allowed distance sampling to be conducted on nine days during the 2012 survey, totalling 826 km of track lines. A total of 350 sightings were recorded, to which the primaries contributed 169 observations, comprising a total of 230 porpoises (Figure 3.2). The calculated density for the whole survey area extending the gap-area was 0.786 animals/km² (95% CI 0.498–1.242, CV = 0.235) and the average group size 1.488 animals. The abundance of harbour porpoises within the 51 511 km² survey area was estimated at 40 475 animals (95 % CI 25,614–65,041, CV = 0.235; see Viquerat *et al*., 2014). Both SCANS surveys (Hammond *et al*., 2002; 2013) yielded comparable results for survey areas that were partially covered by the 2012 survey. In 1994, densities were 0.725 animals/km², (CV = 0.34) in block I (Skagerrak, Kattegat and Belt Seas) and 0.101 animals/km², (CV = 0.27) in block X (western part of Baltic). Density in 2005 was 0.280 animals/km² (CV = 0.36) in stratum S, partly covering block I and X but extending to a wider area than the 2012 survey area. It should be noted that these densities had been estimated for areas that are not directly compatible to the survey area in 2012 and have been aggregated over various, independently surveyed blocks of smaller areas. They are thus not directly comparable to the results in 2012, However they hint at the magnitude of population numbers within that general region. As the survey areas potentially covered multiple porpoise populations, further investigations on defining a population boundary and thus identifying a suitable survey area to assess the harbour porpoise population within the Kattegat, Belt Seas and western Baltic are needed.

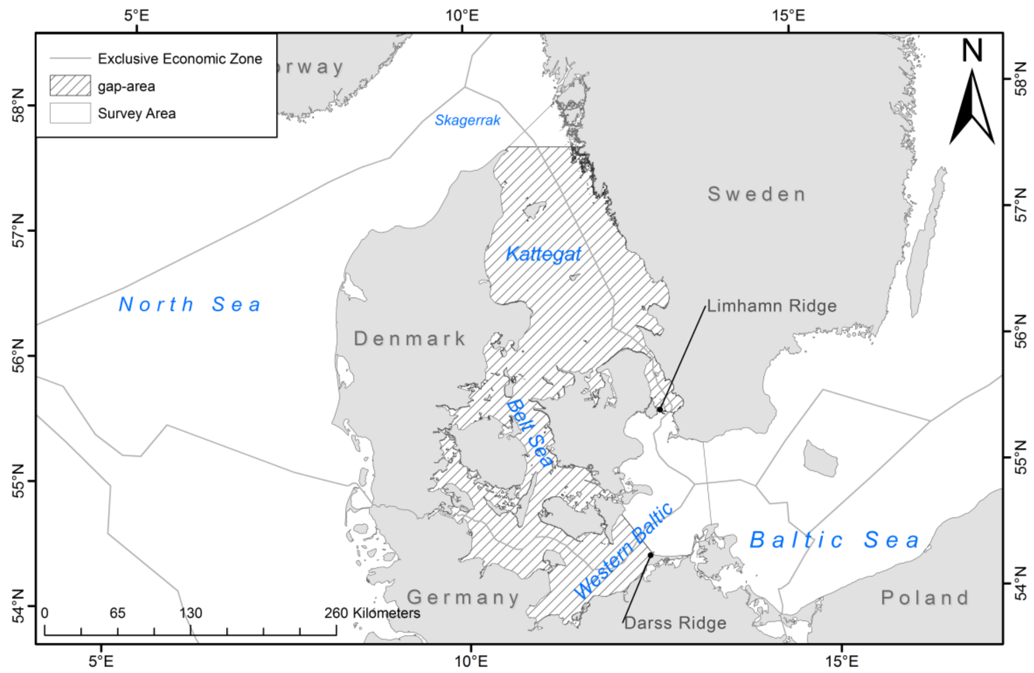

Figure 3.1. Survey area and the gap-area. From Viquerat *et al*., 2014.

The 2012 survey was conducted as part of a five year monitoring period (2011–2015) in Denmark under the EU Habitat Directive and was partly funded by the German Federal Ministry for Food, Agriculture and Consumer Protection. The Danish monitoring programme within the Kattegat, Belt Seas and western Baltic Sea also includes two acoustic surveys (2011 and 2013) and static acoustic monitoring (using CPODs) of Special Areas of Conservation (SACs) in Danish waters. This area could provide a good test region to compare different data collection and analysis methodologies (ship, aerial, acoustic). The German monitoring programme includes regular aerial surveys in summer (every two years; also covering parts of Danish waters in the western Baltic; see Gilles *et al.*, 2011; 2014) and static acoustic monitoring with C-PODs (Gallus *et al.*, 2011; 2014).

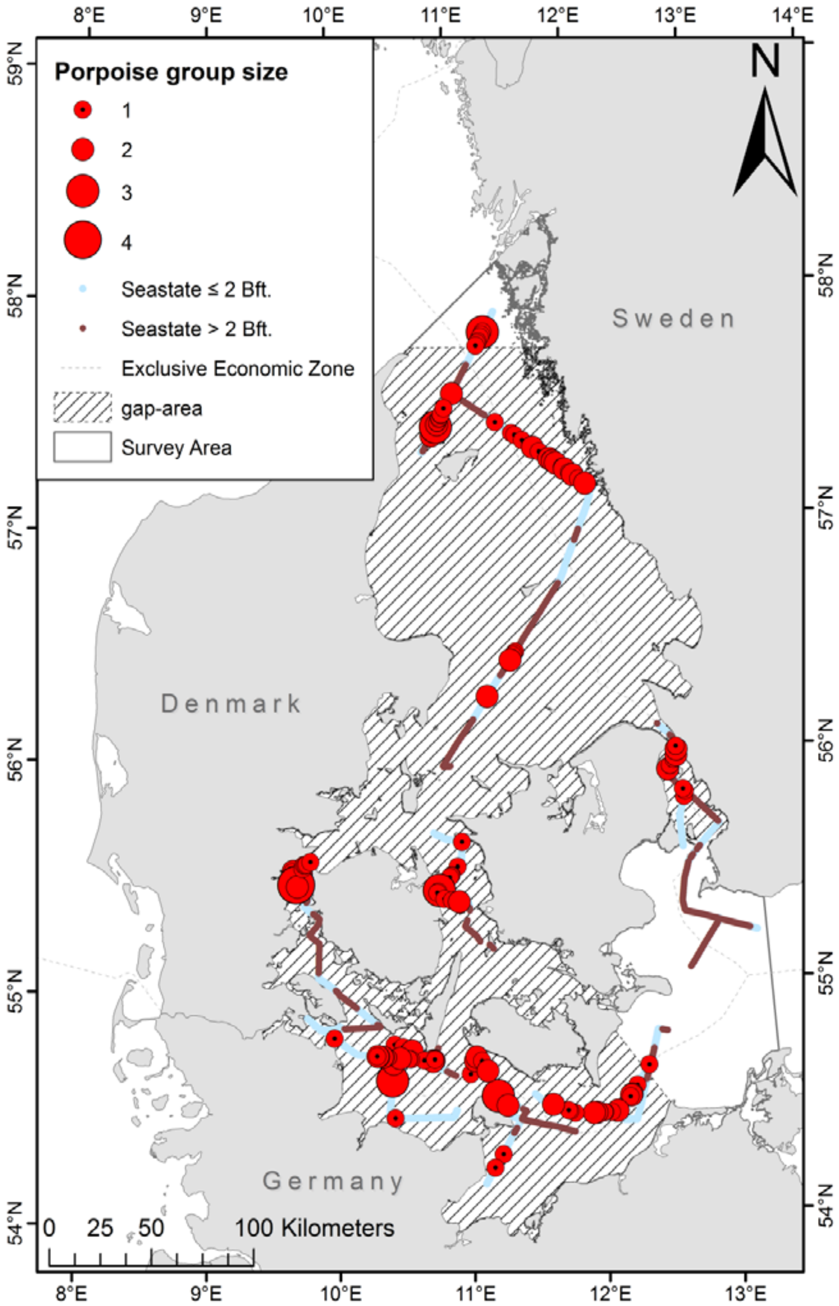

Figure 3.2. Survey results from the 2012 survey. From Viquerat *et al.*, 2014.

3.1.2 Abundance of harbour porpoise in the German North Sea and south-western Baltic Sea

In the framework of the Natura 2000 monitoring programme, dedicated aerial surveys to assess distribution and density of harbour porpoise are being conducted in the German North Sea and western Baltic Sea (Figure 3.3). The results of the surveys in 2013 are presented by Gilles *et al*. (2014). In April and May/June 2013 the area „Borkum Reef Ground" (area D) was surveyed and along 2745 km transect lengths a total of 157 harbour porpoise sightings with 166 individuals (of these one calf) were recorded. The estimated density in area D_West was significantly higher in April, with 1.07 animals/km² (CV=0.29), than in May/June 2013 (0.47 animals/km²; CV=0.44).

In June and July 2013 the area „Sylt Outer Reef" (area C_Nord) was surveyed and along 2832 km transect lengths a total of 464 harbour porpoise sightings with 588 individuals (of these 50 calves) were recorded. Estimated porpoise density was slightly higher in July 2013 (1.75 animals/km², CV=0.25) than in June 2013 (1.52 animals/km², CV=0.26); although differences were not significant.

In July 2013 a survey in the western German Baltic Sea (Kiel Bight and Mecklenburg Bight; area E and F_West) was conducted. Effective survey effort amounted to 1447 km during which 55 harbour porpoise sightings with 78 individuals (of these nine calves) were recorded. Density for the whole study area was estimated to be 0.35 animals/km² (CV=0.32), whereby the density in the Mecklenburg Bight was estimated to be higher than in the Kiel Bight; although statistically not significant.

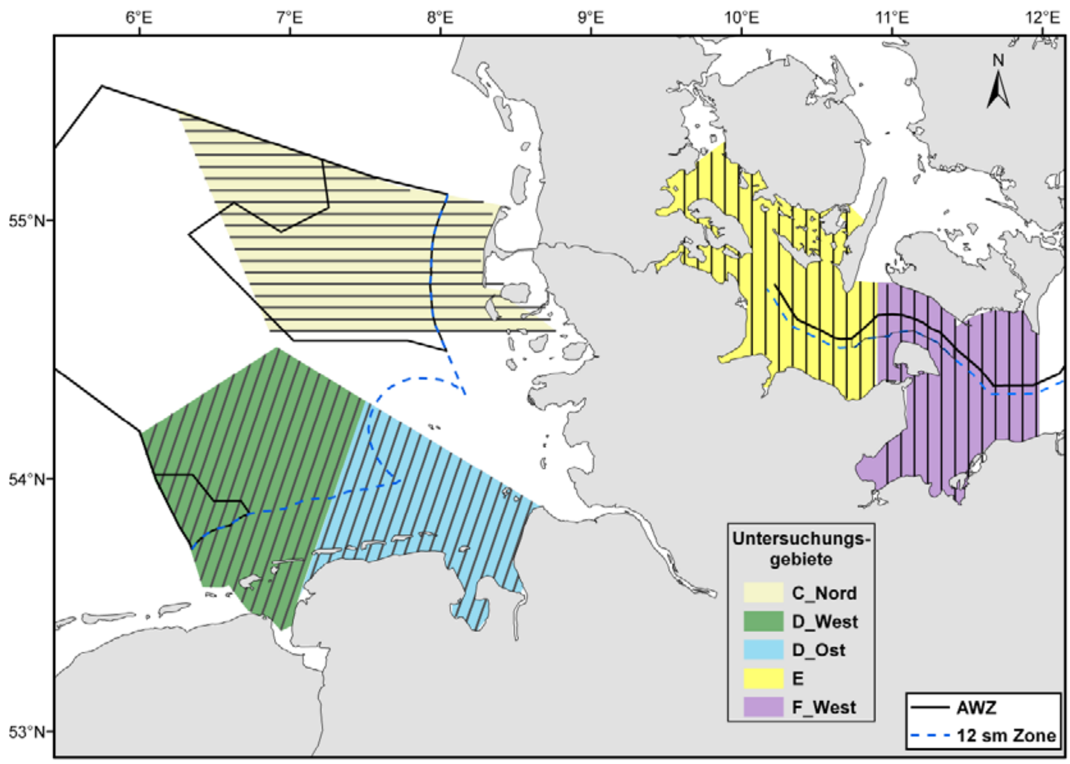

Figure 3.3. Survey blocks in the North and Baltic Sea. Parallel transects are spaced 5 km apart in the North Sea and 6 km in the Baltic Sea. From Gilles *et al.*, 2011b.

3.1.3 Abundance of harbour porpoise in Dutch waters

WGMME (2013) reported on Dutch aerial surveys conducted in March and November 2012, with the aim to assess the seasonal abundance and distribution of harbour porpoise on the Dutch continental shelf (DCS). New information was available this year from surveys in March-April 2013 (Geelhoed *et al.*, 2014b). Surveys were conducted in four strata A-D (Figure 3.4). In total, 197 sightings of 223 individual harbour porpoises were collected on seven survey days between 18 March and 22 April. Porpoise densities varied between 0.47–1.44 animals/km² in the areas A-D (Figure 3.5). The overall density on the entire Dutch continental shelf was 1.07 animals/km². Harbour Porpoises were widely distributed in March with higher densities in area D "Delta". In the northern part of the DSC the distribution seemed patchier with lower densities in the northern part of area B "Offshore" and in area A "Dogger Bank".

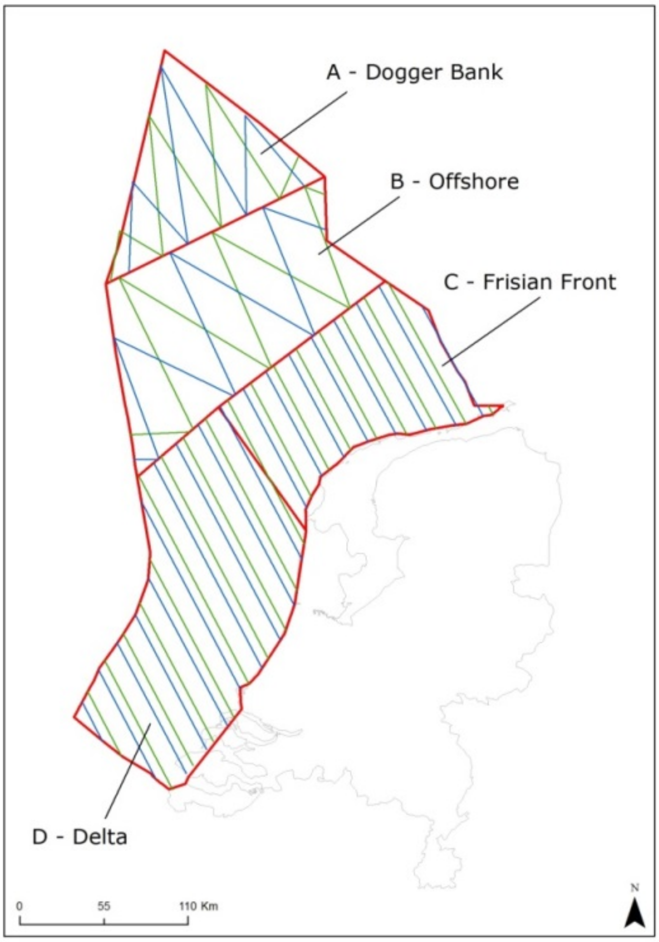

Figure 3.4. Map of the Dutch continental shelf with the track lines in four strata A-D. Colours of track lines indicate whether the lines belong to the same set. From Geelhoed *et al.*, 2013

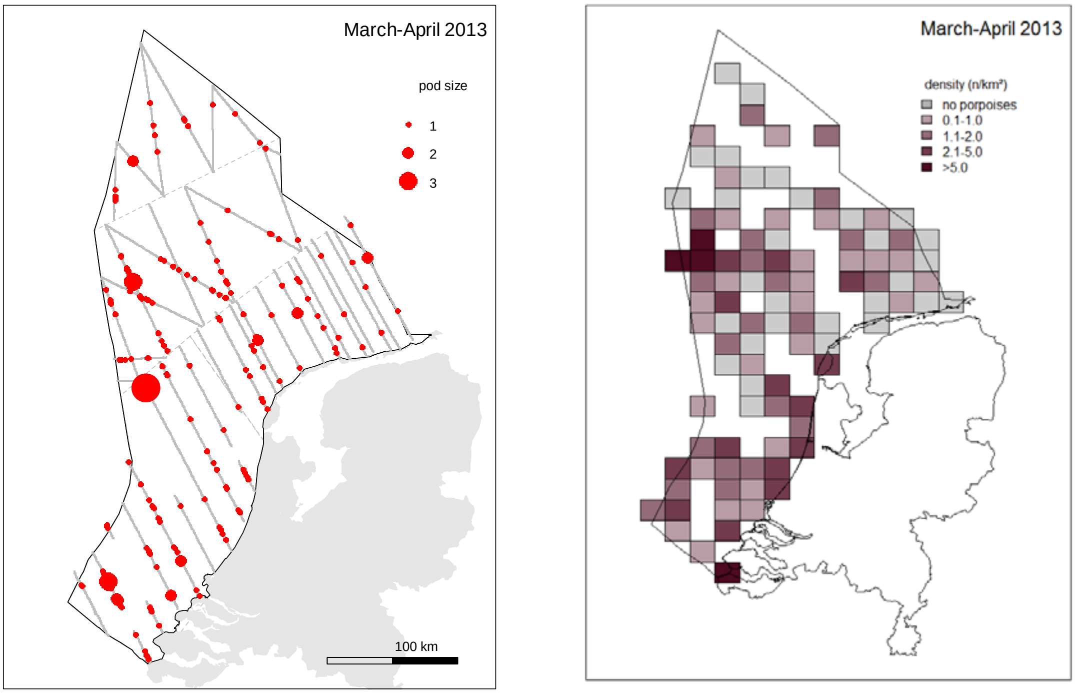

Figure3.5. Left panel: Map of the Dutch continental shelf with the surveyed track lines showing the sightings of harbour porpoise. Right panel: Spring density distribution of harbour porpoises (animals/km²) per 1/9 ICES grid cell, March 2013. Grid cells with low effort (<1 km²) are omitted. From Geelhoed *et al.*, 2014b.

The total numbers of Harbour Porpoises on the Dutch continental shelf (areas A–D) in March were estimated at ca. 63 000 animals (C.I.: 32 000–129 000). Although this number is lower than the population estimate in March 2011 (86 000, C.I.: 49 000–165 000), it is similar to the abundance estimate in March 2012 (66 000, C.I.: 37 000–130 000). However, the confidence intervals of the three estimates greatly overlap and therefore these numbers can be considered of comparable size.

3.1.4 Abundance of harbour porpoise in Belgian waters

Aerial surveys in Belgian waters started in 2008 and continued. Two aerial surveys covering the whole of Belgian waters (1367 km track line) were conducted in 2012, whereas four aerial were performed in 2013 covering 2287 km track line. The results from 2012 and 2013 are presented in Table 3.1 and Figure 3.6. Compared with previous years, the estimate of the density of porpoises in May 2013 was very high, possibly due to the relatively low sea surface temperatures compared to other years during spring 2013. Also in spring 2013, a record number of stranded porpoises was recorded along the Belgian coast (RBINS, unpublished data; record numbers in April, May and June), leading to an overall highest ever recorded number of stranded animals of 127.

Table 3.1. Density estimates for harbour porpoises from aerial line transect surveys covering Belgian waters in 2012 and 2013 (partly from Haelters *et al.*, 2013, partly unpublished data RBINS).

	Effort (km)	N observed animals	Density (animals/km²)
March 2012	696	197	1.63 (1.25–2.11)
October 2012	670	40	0.46 (0.26–0.84)
January 2013	444	49	0.85 (0.45–1.58)
February 2013	572	71	0.95 (0.66–1.37)
May 2013	563	127	1.73 (1.13–2.62)
September 2013	707	57	0.61 (0.41–0.92)

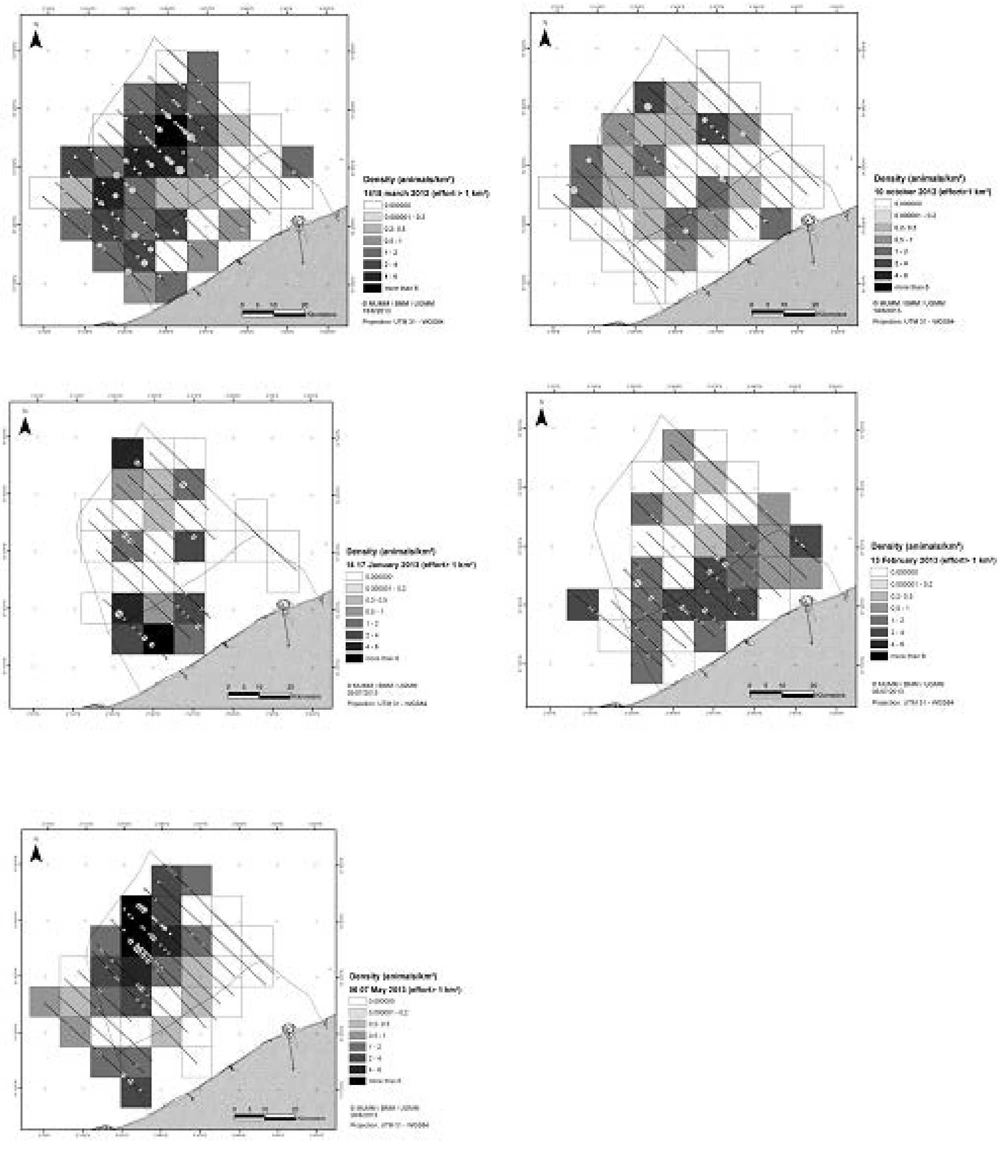

Figure 3.6. Density distribution of harbour porpoises (animals/km²) in Belgian waters in March and October 2012 and January, February and May 2013. From Haelters *et al.*, 2013 and unpublished data.

3.1.5 Distribution of cetaceans in French waters

Progress on Atlas of Mammals in French waters (both overseas territories and mainland France), planned for 2014/2015.

WGMME (2013) reported on dedicated aerial surveys across all waters under French jurisdiction: Suivi Aérien de la Mégafaune Marine (SAMM). These surveys were designed to document distribution of all megafauna (mammals, birds, turtles, and large fish) visible from the air. SAMM does not aim at estimating absolute abundance, but at assessing relative abundance to look at distributional changes in time and space. The programme has two seasonal components: a winter survey was conducted from late November 2011–mid-February 2012 and a summer survey was conducted from mid-May–early August 2012. The second series of survey have been called off for financial reasons. In total survey effort was 48 600 km in winter and 53 200 km in summer, of which ca. 90% was conducted in sea state ≤3 Beaufort in all regions and both seasons (see Figure 3.7). In total about 1500 marine mammal encounters were recorded in winter and 2000 in summer. The preliminary results from SAMM were summarized in the WGMME 2013 report. Analysis of raw data started in 2013 and is ongoing in 2014.

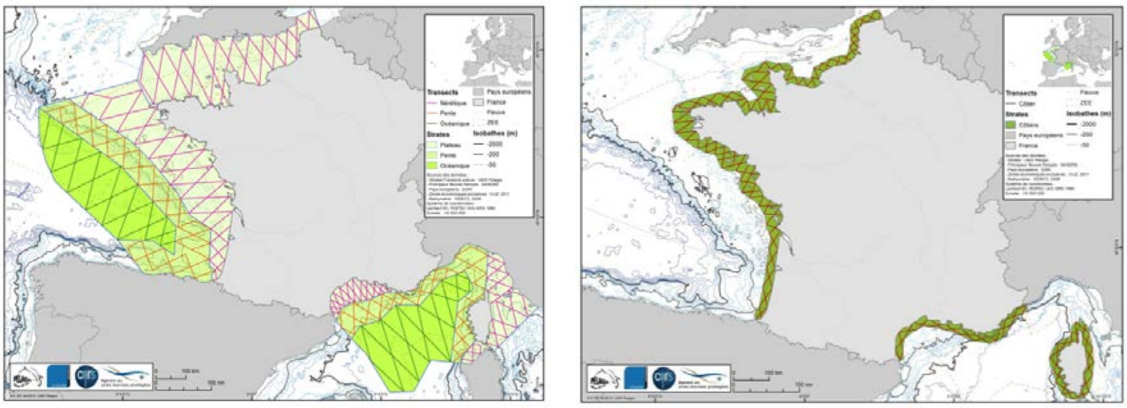

Figure 3.7. Areas covered by and general design of the SAMM survey, showing organization in three strata defined as shelf, slope and oceanic strata (left). A 12nm mile coastal band (right) received additional survey effort in order to increasing resolution where most existing N2000 sites are located (map source ULR-CNRS-AAMP).

3.1.6 Distribution and abundance of cetaceans in continental Portuguese waters

Systematic annual aerial surveys along the coast of continental Portugal were initiated in 2010 with the SafeSea project to investigate the abundance of cetaceans in Portuguese waters, in August/September. These surveys will continue annually until 2014 under the Framework of the MarPro Life+ Project and utilise aerial surveys as well as platforms of opportunity such as oceanographic research vessels and fishing boats. Two types of surveys were conducted; one aimed at seabirds and cetaceans (except Harbour porpoise) extending further offshore, the other aimed at harbour porpoises in the 'near shore' area (see Figure 3.8; Santos *et al*., 2012). The offshore surveys have been conducted in August/September 2010, September 2011 and September 2012 and covered 62 716–74 870 km^2 annually. The near shore harbour porpoise survey was conducted in August/September 2010, September 2011 and 2012, covering 30 033 km^2 each year. However, in the case of 2010 the number of observed animals was not sufficient to produce abundance estimations. Results are presented in Figure 3.9.

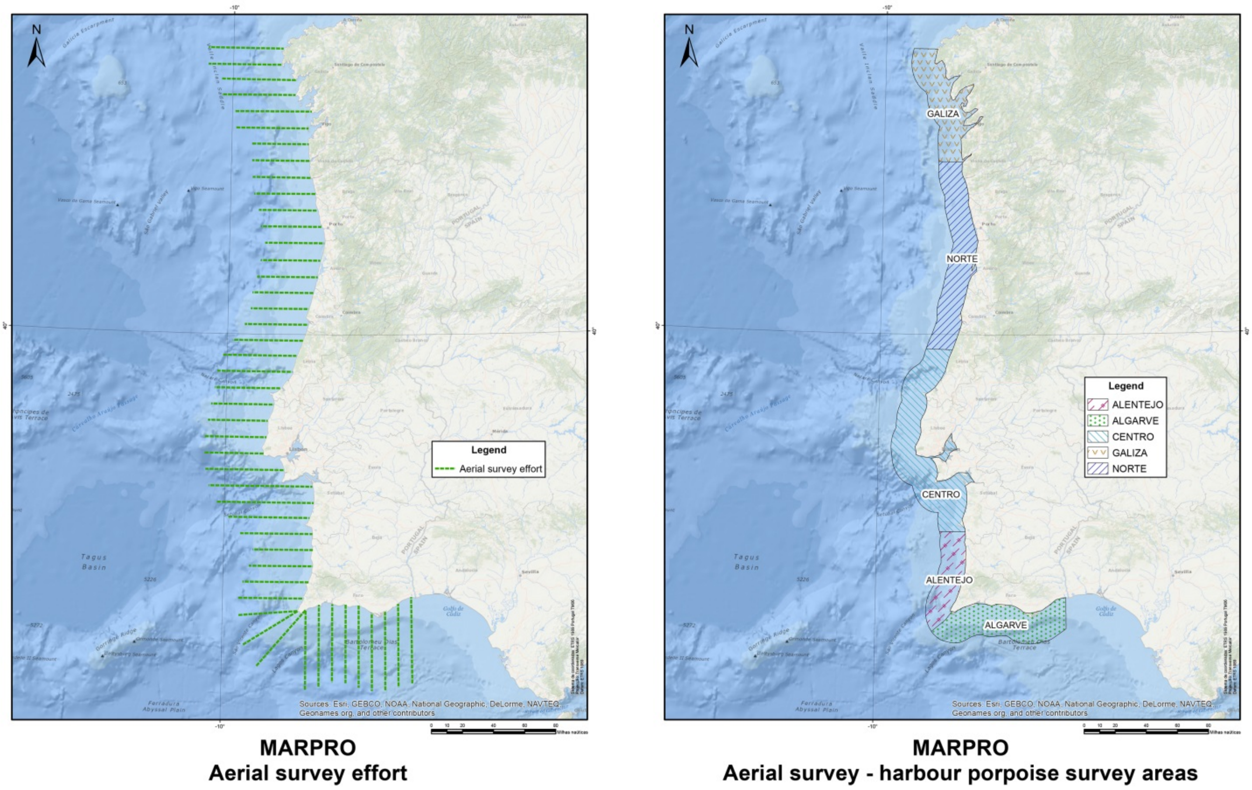

Figure 3.8. Portuguese survey track lines and preliminary results. Left: all species except Harbour porpoise. Right: Harbour porpoise. From Santos *et al.*, 2012.

The dedicated aerial surveys in 2010–2012 made it possible to calculate abundance estimates for minke whale, harbour porpoise, bottlenose dolphin, common dolphin, striped dolphin, and mixed groups of common and striped Dolphin (see Table 3.2). Minke whale were only sighted in 2011 and 2012, resulting in abundance estimates of 2919 (C.I.: 1247–6834) and 3248 (C.I.: 1113–9477), respectively. In 2011 and 2012 abundance estimates for harbour porpoise were 1691 (C.I.: 406–7049) and 3593 (C.I.: 856–6955), respectively. Abundance estimates for common dolphins could be calculated for all three years and ranged from 24 055 (C.I.: 12 814–45 158) in 2010 to 45 984 (C.I.: 30 663–69 027) and 48 173 (C.I.: 26 817–86 536) in 2011 and 2012, respectively. Abundance estimates for striped dolphins in 2011 and 2012 were 1771 (C.I.: 513–6106) and 4149 (C.I.: 569–10 970), respectively. The abundance estimates for combined groups of common and striped dolphins, however, ranged between 25 915 (C.I.: 3058–21 9580) in 2010 and 19 305 (C.I.: 1614–230 910) in 2012. For bottlenose dolphin abundance estimates are available for all three years and ranged from 3051 (C.I.: 294–31 666) in 2010 to 4191 (C.I.: 863–20 364) in 2011.

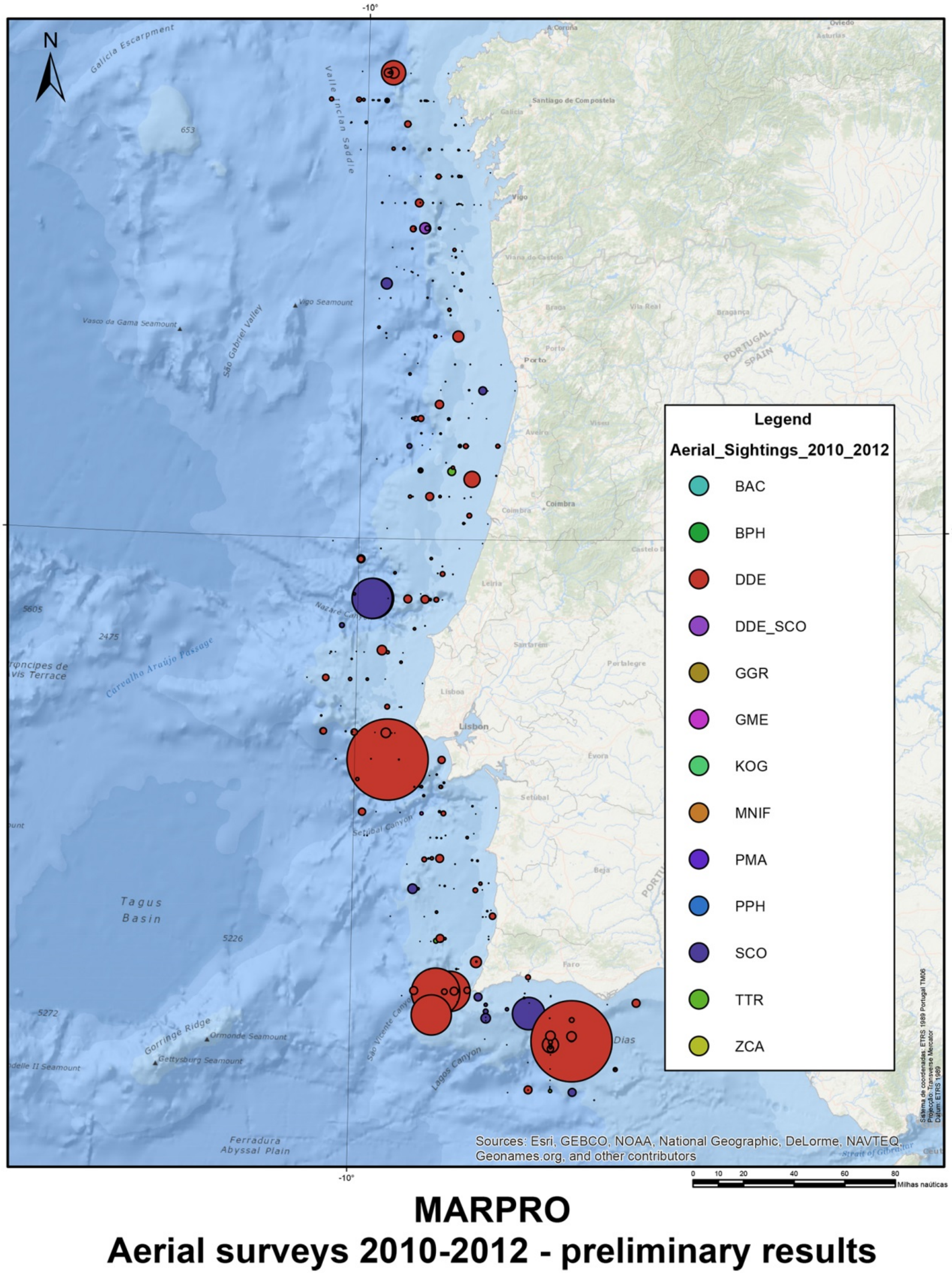

Figure 3.9. Portuguese survey tracks and preliminary results. From Santos *et al.*, 2012.

Table 3.2. Abundance estimates for a few species in Portuguese waters. Correction factors are obtained from Forcada (2004; bottlenose dolphin), Gomez de Segura (2005; striped dolphin) and Witting (2005; minke whale). From Santos *et al.*, 2012.

	Year	N	95%-C.I.	Correction factor	
Minke whale	2010	N/A	-	0.106	
	2011	2919	1247–6834		
	2012	3248	1113–9477		
Harbour porpoise	2010	N/A	-	0.45	
	2011	1691	406–7049		
	2012	3593	856–6955		
Common dolphin	2010	24 055	12 814–45 158	0.676	Assumed to be identical with Striped
	2011	45 984	30 663–69 027		
	2012	48 173	26 817–86 536		
Striped dolphin	2010	N/A	-	0.676	
	2011	2565	760–8662		
	2012	6240	1538–25 319		
Bottlenose dolphin	2010	3051	294–31 666	0.7784	
	2011	4191	863–20 364		
	2012	3935	399–38 806		
Common/striped dolphin	2010	39 858	4931–322 200	0.676	
	2011	N/A	-		
	2012	29 416	2547–339 690		

3.1.7 Abundance of harbour porpoises around the Dogger Bank (North Sea)

Geelhoed *et al*. (2014a) report on a dedicated aerial line transect survey of the Dogger Bank and adjacent areas (Danish, Dutch, German and UK waters) to investigate the importance of this marine feature as summer habitat for marine mammals. This survey repeated the design of Gilles *et al*. (2011a) and comprised eight strata within the 66 768 km^2 study area. On 74 parallel transects planned, a total of 9674 km survey effort (left and right combined) was carried out in moderate to good survey conditions during ten survey days between 20 August and 3 September 2013. In total 619 harbour porpoises were sighted, including 21 calves (Figure 3.10), which resulted in an estimate of 45 177 (C.I. 25 105–84 556) harbour porpoises. Highest porpoise density was found in the northwestern, southern and southwestern parts of the survey area, whereas over the sandbank itself and to the southeast relatively low densities were estimated (Figure 3.11). Additionally to the porpoises, 18 minke whale and 12 white-beaked dolphin, and 35 seals (grey seal and common seal) were recorded. Numbers of these species were too low to calculate densities and abundance estimates.

Compared with a survey of the same study area in summer 2011 (Gilles *et al*., 2011a), the abundance estimate in 2013 was lower: 45 177 (C.I. 25 105–84 556) vs. 116 446 (C.I.

64 423–224 881). The observed distribution showed roughly the same pattern around the Dogger Bank for both years. Compared to 2011, in the areas west of the Dogger Bank porpoise densities in 2013 were higher in the north and lower in the central part.

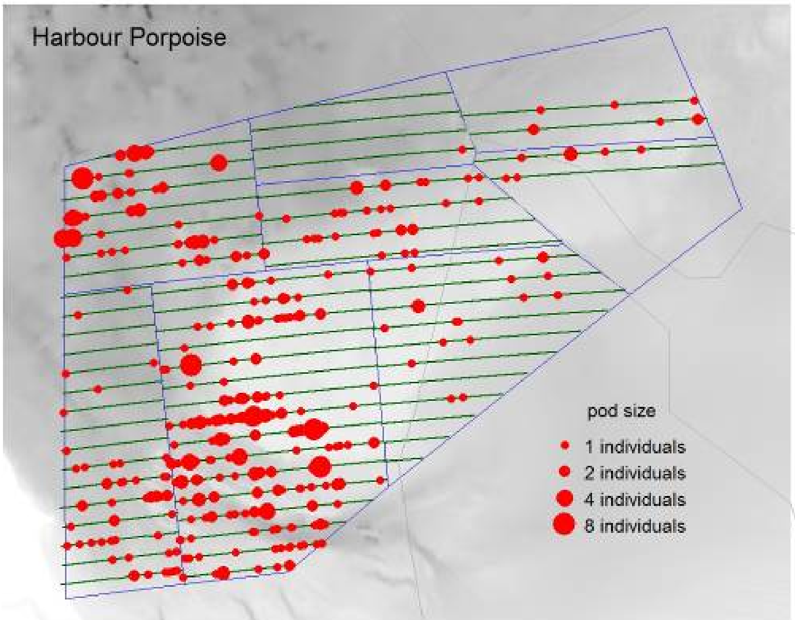

Figure 3.10. Harbour porpoise on effort sightings in the Dogger Bank area in summer 2013. Shading gives an indication of water depth. From Geelhoed *et al.*, 2014a.

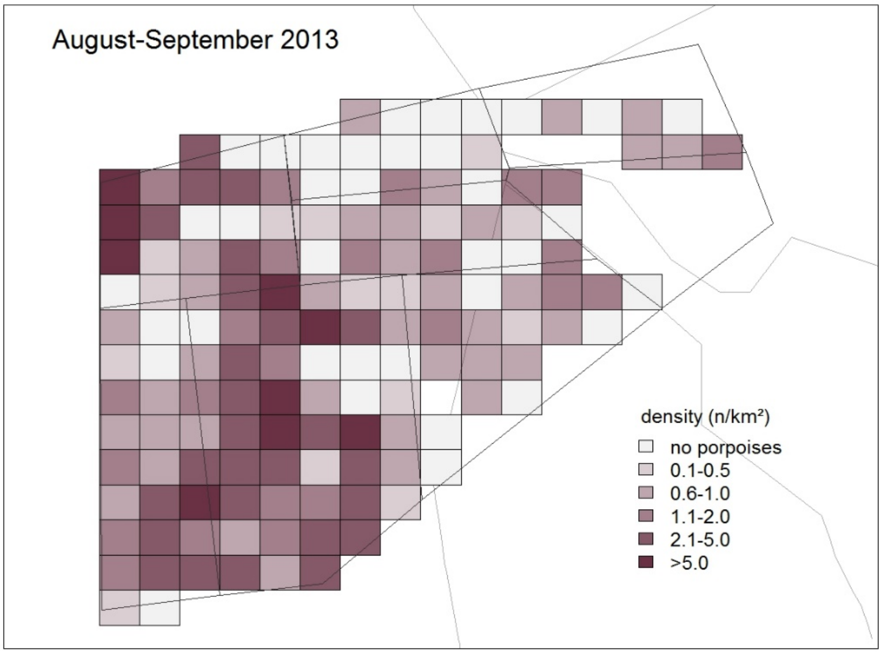

Figure 3.11. Harbour porpoise density (n/km²) per 1/9 ICES grid cell in the Dogger Bank area in summer 2013. From Geelhoed *et al.*, 2014a.

3.1.8 Large-scale cetacean surveys in the European Atlantic

Corrections of the minor errors discovered in the SCANS-II data last year have been made, analyses rerun and the results included in a paper in Biological Conservation (Hammond *et al.*, 2013).

Reanalysis of the combined SCANS-II, CODA and Faroese TNASS data are underway; results are expected to be reported later this year.

3.2 Future surveys

In 2009, WGMME recommended that surveys to estimate absolute abundance such as SCANS-II and CODA continue with frequency of at least between five and ten years and that, if possible, both the shelf and offshore waters should be covered simultaneously (WGMME, 2009).

Preparations for a SCANS-III survey began with an initial preparatory meeting in December 2012. The SCANS-III project will centre on a survey of all cetaceans in shelf and offshore waters in the European Atlantic in summer 2016. Other project elements that are planned to be included are: (a) the creation of a common European database for designed surveys; (b) the collation of data and creation of "risk layers" for bycatch and ship strikes in time and space for cetaceans in the European Atlantic; (c) assessments of risk for all cetacean species based on (a) and (b) and the new abundance data from the survey; (d) the final development and implementation using the new abundance data of a management framework for setting safe limits to bycatch for relevant cetacean species in the European Atlantic; (e) an intensive, focused trial in summer 2015 of different methods of monitoring (including shipboard visual and acoustic, aerial visual and digital, static acoustic) to inform on best practice for monitoring by Member States; (f) a socio-economic assessment of anthropogenic interactions with cetaceans; and (g) capacity building for cetacean conservation.

Preparations are now focusing on developing a proposal to submit for LIFE Nature funding later in 2014 and raising support from EU Member States for that proposal. To inform this proposal, a technical workshop on aerial survey methods is being planned for mid-2014.

WGMME strongly supports the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2016 and **recommends** that it is supported by all range states and by ICES, ASCOBANS and the European Commission. Continuation of these surveys is essential to the accurate estimation of absolute abundance for several species that are required for reporting under the Habitats Directive and the Marine Strategy Framework Directive.

A Trans-North Atlantic Sightings Survey (T-NASS), coordinated through NAMMCO, is being planned for summer 2015 as the latest in a series of such surveys previously conducted in 1987, 1989, 1995, 2001 and 2007. It is expected that this survey will cover a large proportion of central and eastern North Atlantic waters off Norway, Iceland, The Faroe Islands and Greenland. Details have yet to be finalised but the coordinators of T-NASS are working with the coordinators of SCANS-III to ensure that results can be combined in the most efficient way.

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4 ToR B Provide information on abundance, distribution, population structure and incidental capture of marine mammals in the western North Atlantic (North Atlantic right whale, harbour porpoise and white-sided dolphin)

In the US, the Marine Mammal Protection Act as amended in 1994 (MMPA) requires the National Marine Fisheries Service (NMFS) and the US Fish and Wildlife Service (FWS) to prepare stock assessment reports for each stock of marine mammal that occurs in waters under US jurisdiction (Wade and Angliss, 1997). The MMPA requires that each report contain several items, including information on distribution and geographic range, stock structure, abundance and population trends, annual human-caused mortality and serious injury, a description of commercial fisheries that interact with the stock, and an estimate of the potential biological removal level (PBR) for the stock. Generally, stocks designated as Endangered (e.g. North Atlantic Right Whales) under the US Endangered Species Act and non-listed stocks for which new information (e.g. abundance, bycatch, unusual mortality events) is available are updated annually, and draft reports are reviewed by regional Scientific Review Groups, and posted for public comment.

Information presented in this section is a combination of oral presentations from Northeast Fisheries Science Center, Protected Species Branch staff and text extracted from: Waring GT, Josephson E, Maze-Foley K, Rosel, PE, editors. 2013. US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2012. NOAA Tech Memo NMFS NE 223; 419 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543–1026, or online at http://www.nefsc.noaa.gov/nefsc/publications/

4.1 North Atlantic Right Whales (*Eubalaena glacialis*)

4.1.1 Distribution

The western North Atlantic right whale population ranges primarily from calving grounds in coastal waters of the southeastern United States to feeding grounds in New England waters and the Canadian Bay of Fundy, Scotian Shelf, and Gulf of St Lawrence (Figure 4.1). Mellinger *et al.* (2011) reported acoustic detections of right whales near the nineteenth-century whaling grounds east of southern Greenland, but the number of whales and their origin is unknown. However, Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland. In addition, resightings of photographically identified individuals have been made off Iceland, in the old Cape Farewell whaling ground east of Greenland (Hamilton *et al.*, 2007), northern Norway (Jacobsen *et al.*, 2004), and the Azores (Silva *et al.*, 2012). The September 1999 Norwegian sighting represents one of only two published sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not currently well described. The few published records from the Gulf of Mexico (Moore and Clark, 1963; Schmidly *et al.*, 1972) represent either distributional anomalies, occasional wanderings of individual animals, or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern United States. Whatever the case, the location of much of the population is unknown during winter.

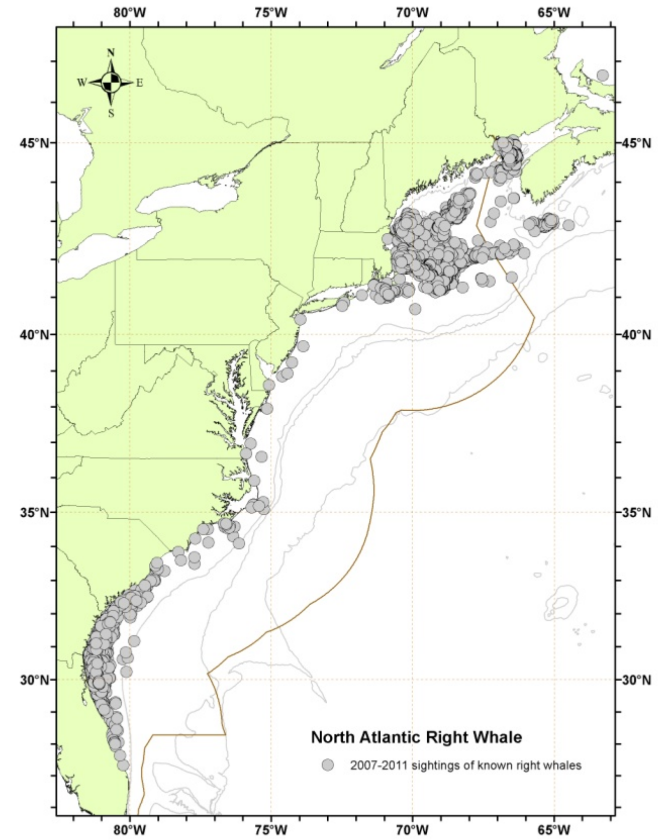

Figure 4.1. Distribution of sightings of known North Atlantic right whales, 2007–2011. Isobaths are the 100 m, 1000 m and 4000 m depth contours.

Research results suggest the existence of six major habitats or congregation areas for western North Atlantic right whales: the coastal waters of the southeastern United States; the Great South Channel; Georges Bank/Gulf of Maine; Cape Cod and Massachusetts Bays; the Bay of Fundy; and the Scotian Shelf. However, movements within and between habitats are extensive and the area off the Mid-Atlantic States is an important migratory corridor. In 2000, one whale was photographed in Florida waters on 12 January, then again eleven days later (23 January) in Cape Cod Bay, less than a month later off Georgia (16 February), and back in Cape Cod Bay on 23 March, effectively making the round-trip migration to the Southeast and back at least twice during the winter season (Brown and Marx, 2000). Results from satellite tags clearly indicate that sightings separated by perhaps two weeks should not necessarily be assumed to indicate a stationary or resident animal. Instead, telemetry data have shown rather lengthy and somewhat distant excursions, including into deep water off the continental shelf (Mate *et al.*, 1997; Baumgartner and Mate, 2005). Systematic surveys conducted off the coast of North Carolina during the winters of 2001 and 2002 sighted eight calves, suggesting the calving grounds may extend as far north as Cape Fear. Four of the calves were not sighted by surveys conducted further south. One of the females photographed was new to researchers, having effectively eluded identification over the period of its maturation (McLellan *et al.*, 2004). There is also at least one recent case of a calf apparently being born in the Gulf of Maine (Patrician *et al.*, 2009).

4.1.2 Stock structure

Genetic analyses based upon direct sequencing of mitochondrial DNA (mtDNA) have identified seven mtDNA haplotypes in the western North Atlantic right whale, including hetroplasmy that led to the declaration of the 7th haplotype (Malik *et al.*, 1999; McLeod and White, 2010). Schaeff *et al.* (1997) compared the genetic variability of North Atlantic and southern right whales (*E. australis*), and found the former to be significantly less diverse, a finding broadly replicated by Malik *et al.* (2000). The low diversity in North Atlantic right whales might be indicative of inbreeding, but no definitive conclusion can be reached using current data. Additional work comparing modern and historic genetic population structure, using DNA extracted from museum and archaeological specimens of baleen and bone, has suggested that the eastern and western North Atlantic populations were not genetically distinct (Rosenbaum *et al.*, 1997; 2000). However, the virtual extirpation of the eastern stock and its lack of recovery in the last hundred years strongly suggest population subdivision over a protracted (but not evolutionary) time-scale. Genetic studies concluded that the principal loss of genetic diversity occurred prior to the 18th century (Waldick *et al.*, 2002). However, revised conclusions that nearly all the remains in the North American Basque whaling archaeological sites were bowhead whales and not right whales (Rastogi *et al.*, 2004) contradict the previously held belief that Basque whaling during the 16th and 17th centuries was principally responsible for the loss of genetic diversity.

High-resolution (i.e. using 35 microsatellite loci) genetic profiling has been completed for 66% of all North Atlantic right whales identified through 2001. This work has improved our understanding of genetic variability, number of reproductively active individuals, reproductive fitness, parentage and relatedness of individuals (Frasier *et al.*, 2007).

One emerging result of the genetic studies is the importance of obtaining biopsy samples from calves on the calving grounds. Only 60% of all known calves are seen with their mothers in summering areas, when their callosity patterns are stable enough to reliably make a photo-ID match later in life. The remaining 40% are not seen on a known summering ground. Because the calf's genetic profile is the only reliable way to establish parentage, if the calf is not sampled when associated with its mother early on, then it is not possible to link it with a calving event or to its mother, and information such as age and familial relationships is lost. From 1980 to 2001, there were 64 calves born that were not sighted later with their mothers and thus unavailable to provide age-specific mortality information (Frasier *et al.*, 2007). An additional interpretation of paternity analyses is that the population size may be larger than was previously thought. Fathers for only 45% of known calves have been genetically determined. However, genetic profiles were available for 69% of all photo-identified males (Frasier, 2005). The conclusion was that the majority of these calves must have different fathers that cannot be accounted for by the unsampled males and the population of males must be larger (Frasier, 2005). This inference of additional animals that have never been captured photographically and/or genetically suggests the existence of habitats of potentially significant use that remain unknown. Since 2006, collaborators have sampled approximately 66% of the calves detected in the wintering grounds.

4.1.3 Abundance

The western North Atlantic minimum stock size is based on a census of individual whales identified using photo-identification techniques. A review of the photo-ID recapture database as it existed on 29 October 2012 indicated that 465 individually recognized whales in the catalogue were known to be alive during 2011. This number represents a minimum population size. This is a direct count and has no associated coefficient of variation.

Previous estimates using the same method with the added assumption that whales seen within the previous seven years were still alive have resulted in counts of 295 animals in 1992 (Knowlton *et al*., 1994) and 299 animals in 1998 (Kraus *et al*., 2001). An International Whaling Commission (IWC) workshop on status and trends of western North Atlantic right whales gave a minimum direct-count estimate of 263 right whales alive in 1996 and noted that the true population was unlikely to be substantially greater than this (Best *et al*., 2001).

4.1.4 Current population trend

The population growth rate reported for the period 1986–1992 by Knowlton *et al.* (1994) was 2.5% (CV=0.12), suggesting that the stock was showing signs of slow recovery, but that number may have been influenced by discovery phenomenon as existing whales were recruited to the catalogue. Work by Caswell *et al.* (1999) suggested that crude survival probability declined from about 0.99 in the early 1980s to about 0.94 in the late 1990s. The decline was statistically significant. Additional work conducted in 1999 was reviewed by the IWC workshop on status and trends in this population (Best *et al*., 2001); the workshop concluded based on several analytical approaches that survival had indeed declined in the 1990s. Although capture heterogeneity could negatively bias survival estimates, the workshop concluded that this factor could not account for the entire observed decline, which appeared to be particularly marked in adult females. Another workshop was convened by NMFS in September 2002, and it reached similar conclusions regarding the decline in the population (Clapham, 2002). At the time, no one examined the early part of the recapture series for excessive retrospective recaptures which had the potential to positively bias survival as the catalogue was being developed.

An increase in mortality in 2004 and 2005 was cause for serious concern (Kraus *et al*., 2005). Calculations based on demographic data through 1999 (Fujiwara and Caswell, 2001) indicated that this mortality rate increase would reduce population growth by approximately 10% per year (Kraus *et al*., 2005). Of those mortalities, six were adult females, three of which were carrying near-term foetuses. Furthermore, four of these females were just starting to bear calves, losing their complete lifetime reproduction potential. Strong evidence of flat or negative growth exists in the time-series of minimum number alive during 1998–2000, which coincided with very low calf production in 2004. However, the population has continued to grow since that apparent interval of decline (Figure 4.2).

Examination of the minimum number alive population index calculated from the individual sightings database, as it existed on 25 October 2013, for the years 1990–2011 (Figure 4.2) suggests a positive and slowly accelerating trend in population size. These data reveal a significant increase in the number of catalogued whales with a geometric mean growth rate for the period of 2.8%.

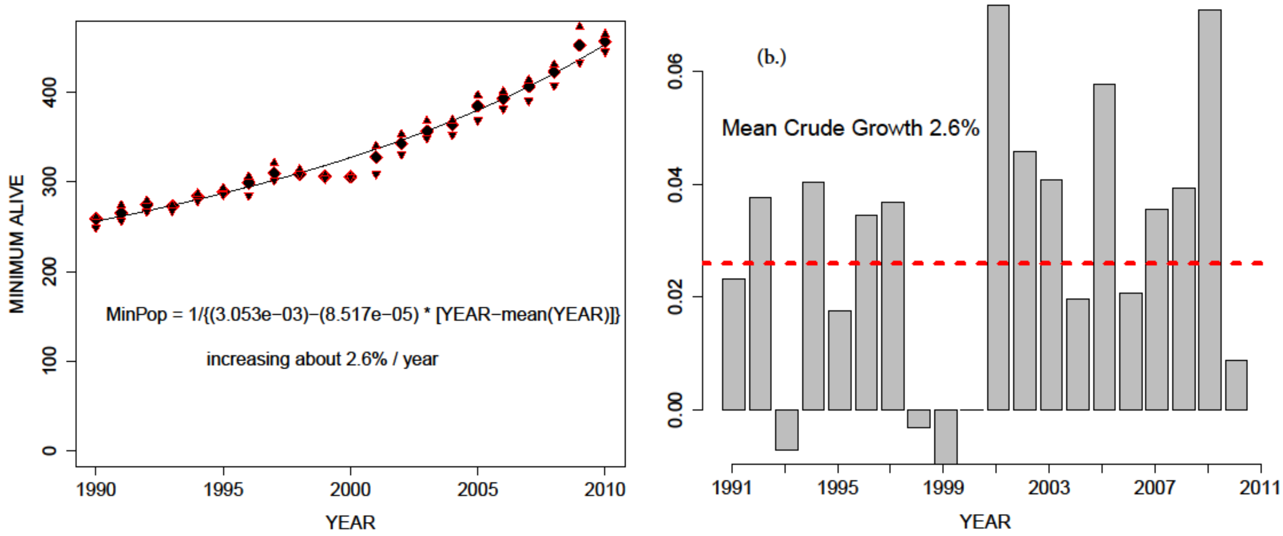

Figure 4.2. Minimum number alive (a) and crude annual growth rate (b) for catalogued North Atlantic right whales. Minimum number (N) of catalogued individuals known to be alive in any given year includes all whales known to be alive prior to that year and seen in that year or subsequently plus all whales newly catalogued that year. Catalogued whales may include some but not all calves produced each year. Bracketing the minimum number of catalogued whales is the number without calves (below) and that plus calves above, the latter which yields Nmin for purposes of stock assessment. Mean crude growth rate (dashed line) is the exponentiated mean of log$_e$ [(N_{t+1}-N_t)/N_t] for each year (t).

4.1.5 Entanglement and ship strike serious injury and mortality

Entanglement in fishing gear and ship strikes are major causes of serious injury and mortality in right whales (Henry *et al*., 2012). For the period 2008 through 2012, the minimum rate of annual human-caused mortality and serious injury to right whales from both US and Canadian waters, averaged 4.75 per year. This is derived from two components: 1) incidental fishery entanglement records at 3.85 per year, and 2) ship strike records at 0.9 per year. Of the 19 reported fisheries entanglements from US waters during this 5-year time period that were classified as serious injury or mortality, 4 were reported before the Atlantic Large Whale Take Reduction Plan's

(http://www.nmfs.noaa.gov/pr/interactions/trt/teams.htm#alw) sinking-groundline rule went into effect in April 2009, and 15 were reported after enactment of the rule. All four of the reported ship strike serious injury and mortalities from US waters during this 5-year time period were after the speed limit rule which went into effect in December 2009, although none were known to occur in areas where the rule mandates speed restrictions.

4.2 Atlantic white-sided dolphins (*Lagenorhynchus acutus)*

4.2.1 Distribution

White-sided dolphins are found in temperate and subpolar waters of the North Atlantic, primarily in continental shelf waters to the 100 m depth contour. In the western North Atlantic the species inhabits waters from central West Greenland to North Carolina (about 35°N) and perhaps as far east as 29°W in the vicinity of the Mid-

Atlantic Ridge (Evans, 1987; Hamazaki, 2002; Doksaeter *et al.*, 2008; Waring *et al.*, 2008).

4.2.2 Stock structure

Distribution of sightings, strandings and incidental takes suggest the possible existence of three stock units: Gulf of Maine, Gulf of St Lawrence and Labrador Sea stocks (Palka *et al.*, 1997; Figure 4.3). Evidence of a separation between the population in the southern Gulf of Maine and the Gulf of St Lawrence population comes from the reduced density of summer sightings along the Atlantic side of Nova Scotia. This was reported in Gaskin (1992), is evident in Smithsonian stranding records and in Canadian/west Greenland bycatch data (Stenson *et al.*, 2011) and was obvious during summer abundance surveys that covered waters from Virginia to the Gulf of St Lawrence and during the Canadian component of the Trans-North Atlantic Sighting Survey in the summer of 2007 (Lawson and Gosselin, 2009). White-sided dolphins were seen frequently in Gulf of Maine waters and in waters at the mouth of the Gulf of St Lawrence, but only a relatively few sightings were recorded between these two regions. This trend seems to be less obvious in recent years, since 2007.

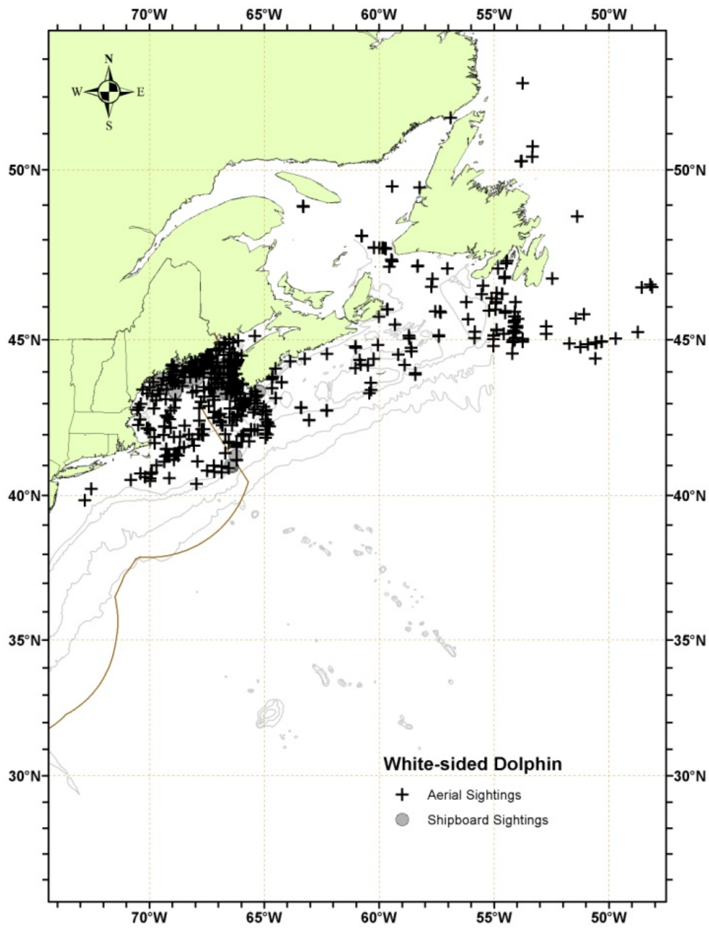

Figure 4.3. Distribution of white-sided dolphin sightings from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, and 2011, and DFO's 2007 TNASS survey. Isobaths are the 100 m, 1000 m and 4000 m depth contours

A study of genetic differentiation among North Atlantic populations did not show any differences between Western North Atlantic populations and those from the eastern North Atlantic (Banguera-Hinestroza *et al*., 2009; 2010). In addition, the analysis of two temporally unrelated populations in the western North Atlantic (from Cape Cod and the Gulf of Maine) showed evidence of the following hypothesis: 1) the existence of several stocks of white-sided dolphins in the western North Atlantic, as previously suggested by Palka *et al*., 1997); and 2) the existence of a refugial population in the Gulf of Maine (Wares, 2002; Adams *et al*., 2006).

4.2.3 Abundance

Abundance estimates of white-sided dolphins from various portions of their range are available from: spring, summer and autumn 1978–1982; July–September 1991–1992; June–July 1993; July–September 1995; July–August 1999; August 2002; June–July 2004; August 2006; July–August 2007; and July–August 2011. The best available current abundance estimate for white-sided dolphins in the Gulf of Maine/Bay of Fundy region is the result of the 2011 survey: 48 819 (CV= 0.61).

An abundance estimate of 24 422 (CV=0.49) white-sided dolphins was generated from the Canadian Trans-North Atlantic Sighting Survey in July–August 2007. This aerial survey covered waters from northern Labrador to the Scotian Shelf, providing full coverage of the Atlantic Canadian coast (Lawson and Gosselin, 2009). The abundance estimates from this survey have been corrected for perception and availability bias, when possible. In general this involved correcting for perception bias using mark-recapture distance sampling (MRDS), and correcting for availability bias using dive/surface times, as reported in the literature, and the Laake *et al*. (1997) analysis method (Lawson and Gosselin, 2011).

4.2.4 Fishery bycatch

Since the mid-1970s, incidental take of Atlantic white-sided dolphins has been observed in various fisheries off the northeast US NMFS observers in the Atlantic foreign mackerel fishery reported 44 takes in fishing activities in the continental shelf and continental slope waters between March 1977 and December 1991 (Waring *et al*., 1990; NMFS unpublished data). Of these animals, 96% were taken in the Atlantic mackerel fishery. This total included nine documented takes by US vessels involved in joint-venture (JV) fishing operations in which US captains transferred their catches to foreign processing vessels.

During 1991 to 1998, two white-sided dolphins were observed taken in the Atlantic pelagic drift gillnet fishery, both in 1993. Estimated annual fishery-related mortality and serious injury (CV in parentheses) was 4.4 (.71) in 1989, 6.8 (.71) in 1990, 0.9 (.71) in 1991, 0.8 (.71) in 1992, 2.7 (0.17) in 1993 and 0 in 1994, 1995, 1996, and 1998. There was no fishery during 1997 and the fishery was permanently closed in 1999.

A US JV mid-water (pelagic) trawl fishery for Atlantic herring was conducted during 2001 on Georges Bank from August to December. No white-sided dolphins were incidentally captured. Two white-sided dolphins were incidentally captured in a single mid-water trawl during foreign fishing operations (TALFF). During TALFF fishing operations all nets fished by the foreign vessel are observed.

The mid-Atlantic gillnet fishery occurs year-round from New York to North Carolina and has been observed since 1993. One white-sided dolphin was observed taken in this fishery during 1997. None were observed taken in other years. The estimated annual mortality (CV in parentheses) attributed to this fishery was 0 for 1993 to 1996, 45 (0.82) for 1997, 0 for 1998 to 2001, unknown in 2002 and 0 in 2003–2012.

Three white-sided dolphins were observed taken in northeast mid-water paired trawls. Estimated annual fishery-related mortalities (CV in parentheses) were unknown in 2001–2002, 22 (0.97) in 2003, 0 in 2004, 9.4 (1.03) in 2005, and 0 in 2006–2012.

The Mid-Atlantic bottom-trawl fishery occurs year-round from south of Cape Cod Massachusetts to Cape Hatteras North Carolina and has been observed since 1995. One white-sided dolphin incidental take was observed in 1997, resulting in a mortality estimate of 161 (CV=1.58) animals. No takes were observed from 1998 through 2004 or in 2006 or 2008–2012; one take was observed in 2005 and two in 2007. Although there were no observed takes in the last decade with the exception of 2005 and 2007, a predictive model estimated the following annual fishery-related mortalities (CV in parentheses): 27 (0.17) in 2000, 27 (0.19) in 2001, 25 (0.17) in 2002, 31 (0.25) in 2003, 26 (0.20) in 2004, 38 (0.29) in 2005, 3 (0.53) in 2006, and 2 (1.03) in 2007 (Rossman, 2010).

The northeast sink gillnet fishery operates in the Gulf of Maine, Georges Bank, and southern New England waters**.** Estimated annual white-sided dolphin mortalities (CV in parentheses) in this fishery attributed to the Northeast sink gillnet fishery were 49 (0.46) in 1991, 154 (0.35) in 1992, 205 (0.31) in 1993, 240 (0.51) in 1994, 80 (1.16) in 1995, 114 (0.61) in 1996 (Bisack 1997), 140 (0.61) in 1997, 34 (0.92) in 1998, 69 (0.70) in 1999, 26 (1.00) in 2000, 26 (1.00) in 2001, 30 (0.74) in 2002, 31 (0.93) in 2003, 7 (0.98) in 2004, 59 (0.49) in 2005, and 41 (0.71) in 2006. New serious injury criteria were applied to all observed interactions retroactive back to 2007 (Waring *et al.*, in press; NOAA, 2012). Estimated fishery-related serious injury and mortality were 0 in 2007, 81 (0.57) in 2008, 0 in 2009, 66 (0.90) in 2010, 18 (0.43) in 2011, and 9 (0.92) in 2012 (Orphanides, 2013; Hatch and Orphanides, in press).

The northeast bottom-trawl fishery operates year-round and primarily on the continental shelf and is distributed throughout the Gulf of Maine, Georges Bank and Southern New England Regions. White-sided dolphin mortalities documented between 1991 and 2006 in this fishery were 1 during 1992, 0 in 1993, 2 in 1994, 0 in 1995–2001, 1 in 2002, 12 in 2003, 16 in 2004, 47 in 2005, and 4 in 2006, 2 in 2007. New serious injury criteria were applied to all observed interactions retroactive back to 2007 (Waring *et al.*, in press; NOAA, 2012). Total observed serious injury and mortality were two in 2007, three in 2008, 31 in 2009, ten in 2010, 49 in 2011, and nine in 2012. Estimated annual fishery-related mortalities (CV in parentheses) were 110 (0.97) in 1992, 0 in 1993, 182 (0.71) in 1994, 0 in 1995–1999, 137 (0.34) in 2000, 161 (0.34) in 2001, 70 (0.32) in 2002, 216 (0.27) in 2003, 200 (0.30) in 2004, 213 (0.28) in 2005, and 40 (0.50) in 2006. Estimated fishery-related serious injury and mortality were 29 (0.66) in 2007, 13 (0.57) in 2008, 168 (0.28) in 2009, 36 (0.32) in 2010, 138 (0.24) in 2011 and 27 (0.47) in 2012.

4.2.5 Mid-Atlantic mid-water trawl fishery (including pair trawl)

In March 2005, five white-sided dolphins were observed taken in paired trawls targeting mackerel that were off Virginia. In February 2006, three animals were observed taken in mackerel paired mid-water trawls north of Hudson Canyon. In March 2007, an animal was observed taken in a mackerel single mid-water trawl near Hudson Canyon. In January and February 2008 three animals were observed in herring single mid-water trawls north of Hudson Canyon. In March 2009 an animal was observed in a pair trawl targeting mackerel south of Hudson Canyon. No white-sided dolphin interactions with this fishery were observed in 2010–2012. Estimated annual fishery-related mortalities (CV in parentheses) were unknown in 2001–2002, 0 in 2003, 22 (0.99) in 2004, 58 (1.02) in 2005, 29 (0.74) in 2006, 12 (0.98) in 2007, 15 (0.73) in 2008, four (0.92) in 2009, and 0 in 2010–2012.

4.2.6 Canada

There is little information available that quantifies fishery interactions involving white-sided dolphins in Canadian waters. Two white-sided dolphins were reported caught in groundfish gillnet sets in the Bay of Fundy during 1985 to 1989, and nine were reported taken in West Greenland between 1964 and 1966 in the now non-operational salmon driftnets (Gaskin, 1992). Several (number not specified) were also taken during the 1960s in the now non-operational Newfoundland and Labrador groundfish gillnets. A few (number not specified) were taken in an experimental drift gillnet fishery for salmon off West Greenland which took place from 1965 to 1982 (Read, 1994).

Hooker *et al*. (1997) summarized bycatch data from a Canadian fisheries observer programme that placed observers on all foreign fishing vessels operating in Canadian waters, on 25–40% of large Canadian fishing vessels (greater than 100 feet long), and on approximately 5% of smaller Canadian fishing vessels. Bycaught marine mammals were noted as weight in kilos rather than by the numbers of animals caught. Thus the number of individuals was estimated by dividing the total weight per species per trip by the maximum recorded weight of each species. During 1991 through 1996, an estimated six white-sided dolphins were observed taken. One animal was from a longline trip south of the Grand Banks (43º10'N 53º08'W) in November 1996 and the other five were taken in the bottom-trawl fishery off Nova Scotia in the Atlantic Ocean; one in July 1991, one in April 1992, one in May 1992, one in April 1993, one in June 1993 and 0 in 1994 to 1996.

Estimation of small cetacean bycatch for Newfoundland fisheries using data collected during 2001 to 2003 (Benjamins *et al*., 2007) indicated that, while most of the estimated 862 to 2228 animals caught were harbour porpoises, a few were white-sided dolphins caught in the Newfoundland nearshore gillnet fishery and offshore monkfish/skate gillnet fisheries.

Stenson *et al*. (2011) examined bycatch of small cetaceans from the 1965–2001 experimental salmon driftnet fishery conducted off southern Newfoundland, Labrador Sea and West Greenland. Fifty-five white-sided dolphins were taken in the West Greenland Labrador Sea, twelve in the Newfoundland Basin and four on the Southern Grand Banks.

4.3 Harbour Porpoise (*Phocena phocena phocena*)

In the western North Atlantic harbour porpoises are distributed from Cape Hatteras, North Carolina to Upernavik, Greenland (Gaskin, 1984; Read, 1999; Teilmann and Dietz, 1998; Lawson *et al*., 2004; Lesage *et al*., 2006), although extralimital strandings have occurred as far south as Florida (Polachek, 1995). Four distinct populations have been identified: Gulf of Maine/Bay of Fundy, Gulf of St Lawrence, Newfoundland-Labrador, and Greenland (Gaskin, 1984; Palka *et al*., 1995; IWC, 1996; Wang *et al*., 1996; Teilmann and Dietz, 1998). In each of these populations, harbour porpoise move into coastal waters during summer, and move offshore in winter, particularly in regions where ice cover is common (Read, 1999).

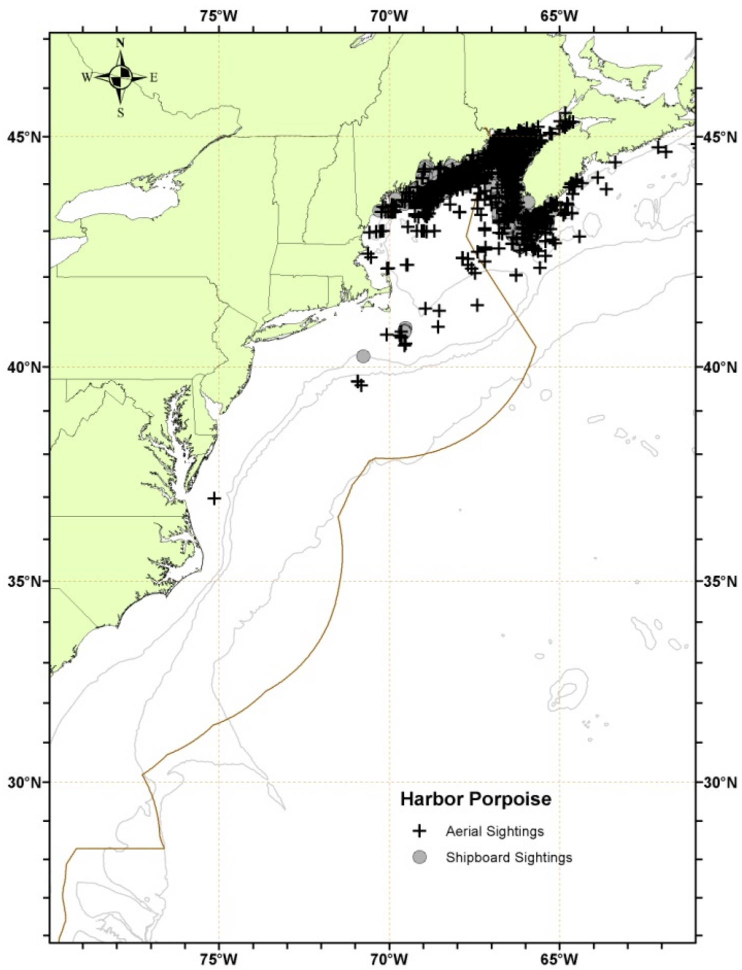

Figure 4.4. Distribution of harbour porpoises from NEFSC and SEFSC shipboard and aerial surveys during the summers of 1995, 1998, 1999, 2002, 2004, 2006, 2007, 2008, 2010, and 2011 and DFO's 2007 TNASS survey. Isobaths are the 100 m, 1000 m, and 4000 m depth contours.

Gulf of Maine/Bay of Fundy: During summer (July to September), harbour porpoises are concentrated in the northern Gulf of Maine and southern Bay of Fundy region, generally in waters less than 150 m deep (Gaskin, 1977; Kraus *et al*., 1983; Palka, 1995a; Palka, 1995b), with a few sightings in the upper Bay of Fundy and on Georges Bank (Palka, 2000) (Figure 4.4). During fall (October–December) and spring (April–June), harbour porpoises are widely dispersed from New Jersey to Maine, with lower densities farther north and south. They are seen from the coastline to deep waters (>1800 m; Westgate *et al*., 1998), although the majority of the population is found over the continental shelf. During winter (January to March), intermediate densities of harbour porpoises can be found in waters off New Jersey to North Carolina, and lower densities are found in waters off New York to New Brunswick, Canada. There does not appear to be a temporally coordinated migration or a specific migratory route to and from the Bay of Fundy region. However, during autumn, several satellite tagged harbour porpoises did favour the waters around the 92 m isobath, which is consistent with observations of high rates of incidental catches in this depth range (Read and Westgate, 1997).

Gulf of St Lawrence: Information in this section was summarized from a more detailed description reported in Lesage *et al*. (2007). Porpoises in the Estuary and Gulf of St Lawrence are suspected to leave the area during winter due to ice cover. During the ice-free period, they were qualified as being moderately abundant by Sergeant *et al.*, (1970). The distribution of harbour porpoises in the Estuary and Gulf of St Lawrence has been characterized using interviews with coastal residents, anecdotal stranding reports, incidental bycatch rates, local observational studies, and systematic surveys. This information indicates that harbour porpoises are ubiquitous in the Estuary and Gulf of St Lawrence during the ice-free period, although seasonal changes in abundance are likely to occur during this period. No information exists on the winter occurrence or distribution of harbour porpoises in the Estuary or Gulf of St Lawrence.

Newfoundland-Labrador: Harbour porpoise are distributed along the east coast of Labrador to Newfoundland, and may be found regularly in deep water (>2000 m) off the continental shelf in the Labrador Sea and Newfoundland Basin (Stenson *et al.*, 2011).

Greenland: The following text is extracted from Teilmann and Dietz (1998). The harbour porpoise is common offshore, inshore and in the fjords in Greenland (Jensen, 1928a,b; Kapel, 1975). The main distribution extends from Sisimiut to Paamiut, with fewer animals seen in the northern and southern municipalities (Lear and Christensen, 1975). Harbour porpoises are rarely seen in east and north Greenland (Vibe, 1971). However, the movements of two satellite-tagged (July 2012) adult females revealed extensive offshore movements and site fidelity to summer feeding grounds (NAMMCO, 2013).

4.3.1 Abundance

The Northeast Fisheries Science Center conducted nine (1991, 1992, 1995, 1999, 2002, 2004, 2006, 2007, and 2011) summer harbour porpoise line-transect abundance surveys in the Gulf of Maine/Bay of Fundy region (citation), sighting surveys were conducted during the summers of 1991, 1992, 1995, 1999, 2002, 2004, 2006, 2007, and 2011. The best current abundance estimate of the Gulf of Maine/Bay of Fundy harbour porpoise stock is the result of the 2011 survey: 79 883 (CV=0.32).

In the Gulf of St Lawrence, systematic surveys conducted in 1995 and 1996 estimated that 36 000 to 125 000 harbour porpoises were summering in the Gulf of St Lawrence (Kingsley and Reeves, 1998). However, the number of porpoises using the Estuary during this period remains uncertain.

In 2007 the Canadian Department of Fisheries and Oceans (DFO) conducted a summer cetacean aerial abundance survey on the Scotian Shelf, in the Gulf of St Lawrence, along the coasts of Newfoundland and Labrador (Lawson and Gosselin, 2009). Harbour porpoise abundance estimates were reported for 1) the combined Scotian Shelf, Cape Breton, and Gulf of St Lawrence survey blocks (N=3667, CV=0.35)), Newfoundland-Labrador blocks (N=1195, CV=0.32).

A large-scale multispecies aerial survey conducted in August–September 2007 and was used to estimate the abundance of harbour porpoises in coastal areas of West Greenland (Hansen and Heide-Jørgensen, 2013). The resultant estimate of the at-surface abundance of harbour porpoises inside the surveyed area corrected for perception bias was 10 314 (CV=0.35). Information from satellite tracking of nine porpoises was used to estimate the proportion of porpoises that can be expected to be outside the survey strata during the survey period. Correcting for this increases the at-surface abundance estimate to 14 129 (CV=0.37) porpoises. Two porpoises tracked from July 2012 through October 2013 provided data on the time spent at the surface during daytime in August–September in both years. The average percentage of time spent at 0 m depth was 5.14% (CV=0.13). Correcting the at-surface abundance estimate for porpoises detected breaking the surface provided a fully corrected abundance estimate of 274 883 (CV=0.39, 95% CI 130 974–576 909) harbour porpoises in West Greenland 2007- also see (NAMMCO, 2013) for additional details.

4.3.2 Bycatch

4.3.2.1 Gulf of Maine/Bay of Fundy–USA Fisheries

One harbour porpoise was observed taken in the Atlantic pelagic drift gillnet fishery during 1991–1998; the fishery ended in 1998. This observed bycatch was notable because it occurred in continental shelf edge waters adjacent to Cape Hatteras (Read *et al.*, 1996). Estimated annual fishery-related mortality (CV in parentheses) attributable to this fishery was 0.7 in 1989 (7.00), 1.7 in 1990 (2.65), 0.7 in 1991 (1.00), 0.4 in 1992 (1.00), 1.5 in 1993 (0.34), 0 during 1994–1996 and 0 in 1998. The fishery was closed during 1997.

4.3.2.2 Northeast sink gillnet

In 1990, an observer programme was started by NMFS to investigate marine mammal takes in the Northeast sink gillnet fishery. Bycatch in the northern Gulf of Maine occurs primarily from June to September, while in the southern Gulf of Maine, bycatch occurs from January to May and September to December. Estimated annual bycatch (CV in parentheses) from this fishery was 2900 in 1990 (0.32), 2000 in 1991 (0.35), 1200 in 1992 (0.21), 1400 in 1993 (0.18) (CUD, 1994; Bravington and Bisack, 1996), 2100 in 1994 (0.18), 1400 in 1995 (0.27) (Bisack, 1997), 1200 in 1996 (0.25), 782 in 1997 (0.22), 332 in 1998 (0.46), 270 in 1999 (0.28) (Rossman and Merrick, 1999), 507 in 2000 (0.37), 53 (0.97) in 2001, 444 (0.37) in 2002, 592 (0.33) in 2003, 654 (0.36) in 2004, 630 (0.23) in 2005, 514 (0.31) in 2006, 395 (0.37) in 2007, 666 (0.48) in 2008, 591 (0.23) in 2009, 387 (0.27) in 2010, 273 (0.20) in 2011, and 277 (0.59) in 2012 (Orphanides, 2013, Hatch and Orphanides, in press).

4.3.2.3 Mid-Atlantic gillnet

Before an observer programme was in place for this fishery, Polacheck *et al.* (1995) reported one harbour porpoise incidentally taken in shad nets in the York River, Virginia. In July 1993 an observer programme was initiated in the mid-Atlantic gillnet fishery by NEFSC. Documented bycatch after 1995 was from December to May. Bycatch estimates were calculated using methods similar to that used for bycatch estimates in the Northeast sink gillnet fishery (Bravington and Bisack, 1996; Bisack, 1997. The estimated annual mortality (CV in parentheses) attributed to this fishery was 103 (0.57) for 1995, 311 (0.31) for 1996, 572 (0.35) for 1997, 446 (0.36) for 1998, 53 (0.49) for 1999, 21 (0.76) for 2000, 26 (0.95) for 2001, unknown in 2002, 76 (1.13) in 2003, 137 (0.91) in 2004, 470 (0.51) in 2005, 511 (0.32) in 2006, 58 (1.03) in 2007, 350 (0.75) in 2008, 201 (0.55) in 2009, 259 (0.88) in 2010, 123 (0.41) in 2011 and 63(0.83) in 2012; Orphanides, 2013; Hatch and Orphanides, in press).

4.3.2.4 Northeast bottom trawl

This fishery is active in New England waters in all seasons. Twenty harbour porpoise mortalities were observed in the Northeast bottom-trawl fishery between 1989 and 2008, but many of these are not attributable to this fishery. Decomposed animals are presumed to have been dead prior to being taken by the trawl. One fresh dead take was observed in the Northeast bottom-trawl fishery in 2003, 4 in 2005, one in 2006, one in 2008, and one in 2011. Revised serious injury guidelines were applied for this period (Waring *et al*., in press; NOAA 2012). One serious injury was observed in 2011. Fishery related bycatch rates for years 2008–2012 were estimated using an annual stratified ratio-estimator. These estimates replace the 2008–2010 annual estimates reported in the 2013 stock assessment report that were generated using a different method. The estimated annual mortality (CV in parentheses) attributed to this fishery was 7.2 (0.48) for 2005, 6.5 (0.49) for 2006, 5.6 (0.46) for 2007, 5.6 (0.97) for 2008, 0 for 2009 and 2010, 5.9 (0.71) for 2011, and 0 for 2012. Annual average estimated harbour porpoise mortality and serious injury from the Northeast bottom-trawl fishery from 2008–2012 is 2.3 (0.60) (Table 2).

4.3.2.5 Canadian Fisheries

4.3.2.5.1 Bay of Fundy sink gillnet

During the early 1980s, harbour porpoise bycatch in the Bay of Fundy sink gillnet fishery, based on casual observations and discussions with fishermen, was thought to be low. The estimated harbour porpoise bycatch in 1986 was 94–116 and in 1989 it was 130 (Trippel *et al*., 1996). The Canadian gillnet fishery occurs mostly in the western portion of the Bay of Fundy during summer and early autumn months, when the density of harbour porpoises is highest. Polacheck (1989) reported there were 19 gillnetters active in 1986, 28 active in 1987, and 21 in 1988.

An observer programme implemented in the summer of 1993 provided a total bycatch estimate of 424 harbour porpoises (± 1 SE: 200–648) from 62 observed trips, (approximately 11.3% coverage of the Bay of Fundy trips) (Trippel *et al*., 1996). During 1994, the observer programme was expanded to cover 49% of the gillnet trips (171 observed trips). The bycatch was estimated to be 101 harbour porpoises (95% confidence limit: 80–122), and the fishing fleet consisted of 28 vessels (Trippel *et al*., 1996). During 1995, due to groundfish quotas being exceeded, the gillnet fishery was closed from July 21 to August 31. During the open fishing period of 1995, 89% of the trips were observed, all in the Swallowtail region. Approximately 30% of these observed trips used pingered nets. The estimated bycatch was 87 harbour porpoises (Trippel *et al*., 1996). No confidence interval was computed due to lack of coverage in the Wolves fishing grounds. During 1996, the Canadian gillnet fishery was closed during 20–31 July and 16–31 August due to groundfish quotas. From the 107 monitored trips, the bycatch in 1996 was estimated to be 20 harbour porpoises (DFO, 1998; Trippel *et al*., 1999). Trippel *et al*. (1999) estimated that during 1996, gillnets equipped with acoustic alarms reduced harbour porpoise bycatch rates by 68% over nets without alarms in the Swallowtail area of the lower Bay of Fundy. During 1997, the fishery was closed to the majority of the gillnet fleet during 18–31 July and 16–31 August, due to groundfish quotas. In addition a time-area closure to reduce porpoise bycatch in the Swallowtail area occurred during 1–7 September. From the 75 monitored trips, 19 harbour porpoises were observed taken. After accounting for total fishing effort, the estimated bycatch in 1997 was 43 animals (DFO, 1998). Trippel *et al*. (1999) estimated that during 1997, gillnets equipped with acoustic alarms reduced harbour porpoise bycatch rates by 85% over nets without alarms in the Swallowtail area of the lower Bay of Fundy. The number of monitored trips (and observed harbour porpoise mortalities were 111 (five) for 1998, 93 (three) for 1999, 194 (five) for 2000, and 285 (39) for 2001. The estimated annual mortality estimates were 38 for 1998, 32 for 1999, 28 for 2000, and 73 for 2001 (Trippel and Shepherd, 2004). Estimates of variance are not available.

Since 2002 there has been no observer programme in the Bay of Fundy region, but the fishery is still active. Bycatch for these years is unknown. The annual average of most recent five years with available data (1997–2001) was 43 animals, so this value is used to estimate the annual average for more recent years. However, in 2011 there was little gillnet effort in New Brunswick waters in summer; thus the Canadian porpoise bycatch estimates could have been near zero. The fishermen that sought groundfish went into the mid-Bay of Fundy where traditionally bycatch levels were extremely low. Trippel (pers. comm.) estimated that less than ten porpoise were bycaught in the Canadian fisheries in the Bay of Fundy in 2011. Analysis of port catch records might allow estimation of bycatch for more recent times; however, it would be difficult to also accurately account for the changes in the spatial distribution of the harbour porpoises and fisheries.

4.3.2.5.2 Herring weirs

Harbour porpoises are taken in Canadian herring weirs, but there have been no recent efforts to observe takes in the US component of this fishery. Smith *et al.* (1983) estimated that in the 1980s approximately 70 harbour porpoises became trapped annually and, on average, 27 died annually. In 1990, at least 43 harbour porpoises were trapped in Bay of Fundy weirs (Read *et al.*, 1994). In 1993, after a cooperative programme between fishermen and Canadian biologists was initiated, over 100 harbour porpoises were released alive (Read *et al.*, 1994). Between 1992 and 1994, this cooperative programme resulted in the live release of 206 of 263 harbour porpoises caught in herring weirs. Mortalities (and releases) were eleven (50) in 1992, 33 (113) in 1993, and 13 (43) in 1994 (Neimanis *et al.*, 1995). Since that time, additional harbour porpoises have been documented in Canadian herring weirs: mortalities (releases and unknowns) were five (60, 0) in 1995; two (4, 0) in 1996; 2 (24, 0) in 1997; two (26, 0) in 1998; three (89, 0) in 1999; 0 (13, 0) in 2000 (A. Read, pers. comm.), 14 (296, 0) in 2001, three (46, 4) in 2002, one (26, 3) in 2003, four (53, 2) in 2004; 0 (19, 5) in 2005; two (14, 0) in 2006; three (9, 3) in 2007, 0 (8, 6) in 2008, 0 (3,4) in 2009, one in 2010 (7, 0), 0 (2, 3) in 2011, and 0 (2, 3) in 2012. (Neimanis *et al.*, 2004; H. Koopman and A. Westgate, pers. comm.).

4.3.2.5.3 Gulf of St Lawrence

Stenson (2003) summarized 1989–1990 bycatch data obtained from questionnaires. Bycatch was 2000 harbour porpoises. No new information was identified.

4.3.3 Newfoundland-Labrador

Stenson (2003) summarized Newfoundland-Labrador bycatch data obtained from logbooks, or phone interviews and extrapolated to fishing enterprises for the period 1980–1992. In eastern Newfoundland bycatch ranged from 243 in 1980 to 41 during 1982–1984. In Newfoundland bycatches were 1368 (1980), 1304 (1989), 2852–4416 (1990), and 2283 (1992). Stenson *et al.* (2011) examined bycatch of small cetaceans from the 1965–2001 experimental salmon driftnet fishery conducted off southern Newfoundland, Labrador Sea and West Greenland. Thirty-seven harbour porpoises were taken in the West Greenland Labrador Sea, one in the Labrador nearshore shelf, forty-one in the Newfoundland Basin and six on the Flemish Cap, and four on the Southern Grand Banks. Lawson *et al.* (2004) estimated that approximately 1500 to 3000 harbour porpoises were incidentally caught in the 2002 nearshore cod gillnet fishery in Newfoundland. The 95 percentile range values around the derived estimates were wide (ranging from 126 to 5605) animals. No new information was identified.

4.3.4 Greenland

Summarized from Teilmann and Dietz (1998). The only study aiming to assess the bycatch in a commercial fishery was carried out in 1972, when observers on board eight of the 22 foreign salmon driftnet vessels (12 Danish, four Faroese, six Norwegian) fishing in Greenlandic waters recorded all catches (Lear and Christensen, 1975). The foreign salmon fishery activity lasted from late July to October. An estimated 1500 harbour porpoises were caught by the foreign driftnet fishery in 1972. The large-scale driftnet fishery for salmon was scaled down during 1972±1976, ceased in 1976 and was never resumed (Kapel, 1983). No new information was identified.

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5 ToR C To review the further development of the Bycatch Limit Algorithm framework for determining safe bycatch limits. This work should include harbour porpoise, short-beaked common dolphin and consideration of additional species for which bycatch estimates have been made or suggested as a potential MSFD indicator (e.g. bottlenose dolphin, striped dolphin, harbour seal and grey seal). This should include a comparison with approaches used to assess bycatch in USA

5.1 Introduction

The subject matter of this ToR has been discussed for several years, beginning in WGMME (2008; section 8), resulting in the following ICES advice to the European Commission.

In 2009, ICES advised the European Commission:

> '*that a Catch Limit Algorithm approach is the most appropriate method to set limits on the bycatch of harbour porpoises or common dolphins. In order to use this (or any other) approach, specific conservation objectives must first be specified. In both species improved information on bycatch and the biology of the species would improve the procedure*.'

This was reiterated in 2010:

> '*ICES advised in 2009 of the need for explicit conservation and management objectives for managing interactions between fisheries and marine mammal populations. This advice has not been acted upon. Lacking these objectives, ICES is unable to properly consider the impacts of these interactions in its management advice*.'

In 2013, in the absence of action and in the context of the development of MSFD targets for marine mammal bycatch, WGMME had again strongly recommended that this advice was acted upon. WGMME (2013) made specific recommendations concerning:

1) the need for policy-makers to define the conservation objectives to be used;
2) the need to define the time frame over which the procedure should be modelled to achieve the specified conservation objectives (proposed to be 100 years); and
3) use of simulations within the framework of a bycatch limit algorithm approach to explore whether more than one management area is appropriate in the North Sea for harbour porpoise.

Without decisions on elements (1) and (2), the extent to which a Bycatch Limit Algorithm approach can be developed is limited.

Notwithstanding this, the UK has supported a project, based on previous work under the SCANS-II and CODA projects, to develop further, as far as possible, a bycatch limit algorithm approach to setting safe bycatch limits at a European level for a number of species.

5.2 Development of a bycatch limit algorithm approach

The bycatch limit algorithm approach to setting safe limits to bycatch and other non-natural removals, which was initially developed as part of the SCANS-II project, is based on the framework of the IWC's RMP for commercial whaling (see Section 3.3.3).

The project supported by the UK has made some progress but, in the continued absence of action on conservation objectives, has been unable to use a fully developed procedure to generate safe bycatch limits for those species of interest at a European level. In work so far, the conservation objective has been assumed to be the ASCOBANS interim conservation objective 'to allow populations to recover to and/or maintain 80% of carrying capacity in the long term'. Long term has been defined as 100 years. Maintaining populations at 80% of carrying capacity has been defined as meeting that objective on average; that is, the population level achieved in the long term is 80% of carrying capacity.

One important development that has been made is to introduce a spatial capability into the framework so that safe bycatch limits can be calculated for multiple management areas for a species, as is proposed for bottlenose dolphins, harbour porpoise, grey seal and harbour seal (WGMME 2012; Section 3.1.1). This also provides a framework for simulations to be used to explore how many management areas are needed in situations where this is currently under debate, for example, for harbour porpoise in the North Sea (WGMME 2013; Section 5.4).

The project will report before the end of 2014.

5.3 Comparison of different approaches for setting safe limits to removals

This topic was previously considered in detail by WGMME (2008; section 8.6). Generally, three ways to set safe limits to removals have been adopted for marine mammals. These are (1) a percentage of abundance; (2) the Potential Biological Removal (PBR) approach; and (3) the IWC's Revised Management Procedure (RMP) for commercial whaling and equivalent procedures for indigenous whaling.

5.3.1 Percentage of abundance

The rationale for using a percentage of abundance or population size as an indicator of a safe limit to removals is that it is directly comparable to the maximum net productivity rate, expressed as an annual percentage. Maximum net productivity rate is difficult to estimate and generic values are typically used. A value of 4% is often assumed for cetacean populations; for pinnipeds the value is set higher, e.g. 12% (Wade, 1998).

To take uncertainty into account, percentage indicators of safe limits to removals should be lower than assumed or estimated values of maximum net productivity. In assessments of small cetacean removals, the IWC has used 2% of population size to raise a flag that removals may not be sustainable and 1% as an indicator that more research is desirable (IWC, 1996; 1997).

ASCOBANS has adopted a conservation criterion "to reduce annual bycatch levels of harbour porpoise to levels of 1.7% of the best population estimate" as a mechanism to achieve its interim conservation objective "to allow populations to recover to and/or maintain 80% of carrying capacity in the long term".

As described in WGMME (2012; Section 3.1), this value of 1.7% was calculated during a joint IWC/ASCOBANS workshop (IWC, 2000). Simulation of a simple deterministic population dynamics model with assumed maximum net productivity rate of 4% found that an annual rate of removals of 1.7% would allow a population to achieve 80% of its carrying capacity over a very long time horizon.

5.3.2 Potential Biological Removal (PBR) level

The US Marine Mammal Protection Act (1972) was amended substantially in 1994 to provide, amongst other things, a programme to authorize and control the taking of marine mammals, incidental to commercial fishing operations and the preparation of stock assessments for all marine mammal stocks in waters under US jurisdiction. A stock assessment report includes, amongst other things, a minimum population estimate that is used in the calculation of a Potential Biological Removal (PBR) level for the stock.

A minimum population estimate is defined as an estimate of the number of animals in a stock that is based on the best available scientific information on abundance, incorporating the precision and variability associated with such information; and provides reasonable assurance that the stock size is equal to or greater than the estimate. In practice, this could be a known minimum number of animals (e.g. for the North Atlantic right whale) or the lower 20th percentile of estimated abundance.

The PBR level is defined as the maximum number of animals, not including natural mortality, that may be removed from a stock while allowing that stock to reach or maintain its optimum sustainable population (OSP), defined as being at or above the population level that will result in maximum productivity.

The PBR level is the product of: (1) the minimum population estimate; (2) one-half the maximum theoretical or estimated net productivity rate; and (3) a recovery factor of between 0.1 and 1.0. The form of the PBR equation was determined from simulations (Wade, 1998). The value of the recovery factor is determined by information on the status of the stock. If the stock is declining and/or endangered, it is set to 0.1. If there is no information on status, it is set to 0.5. Values between 0.5 and 1.0 are set for other stocks depending on the information available.

If the estimated level of direct human-caused mortality (due to bycatch and other causes such as ship strikes) exceeds the calculated PBR level the stock is classified as strategic and a take reduction plan is developed and implemented to assist in recovery or prevent depletion of the stock. Take reduction plans are developed by take reduction teams that consist of representatives from the fishing industry, fishery management councils, state and Federal resource management agencies, the scientific community and conservation organizations.

5.3.3 The Revised Management Procedure

The IWC Revised Management Procedure was developed following the decision to implement a pause or "moratorium" in commercial whaling in 1982. The RMP is a method of calculating sustainable removal levels that are consistent with the Commission's objectives for commercial whaling. It was adopted by the Commission in 1994 but has not been implemented. The IWC has also adopted separate management procedures for indigenous or "aboriginal subsistence" whaling which have been implemented and used to set catch limits.

At the heart of the RMP is the "single-stock" Catch Limit Algorithm (CLA). The CLA fits a defined population model to information on catches (including all non-natural removals), population size and a range in maximum net productivity rate to estimate depletion level. Depletion level is defined as the population size for any given year relative to population size prior to exploitation, assumed equivalent to the carrying capacity. A catch control law is then used to calculate a nominal catch limit from the estimated depletion level and productivity rate. If estimated depletion level is less than 0.54, the nominal catch limit is set to zero. The catch control law is "tuned" to achieve a population level of 72% in the long term (100 years).

The RMP recognises that knowledge of population (stock) structure is uncertain and therefore contains rules that define how nominal catch limits set by the CLA are to be used. These rules are based on setting catch limits for defined areas that are considered small enough to ensure that whales of different stocks taken within such areas will be taken in proportion to the relative abundance of those stocks. Catch limits for these "Small Areas" can be combined to give a total catch limit for a larger area in a number of ways, known as RMP "variants".

Which variants are acceptable is determined from the results of a set of case-specific simulations (so-called "Implementation Simulation Trials") developed to capture the uncertainties in knowledge of the species in question. These uncertainties typically relate to lack of knowledge of stock structure but could also be, for example, about catches or bycatches. The acceptability of a variant is determined by how well it meets conservation objectives over the management time period (100 years) as established from a specified set of performance metrics.

Punt and Donovan (2007) give more details about the development and implementation of the RMP. A formal definition of the RMP is given at http://iwc.int/rmpbw.

The bycatch limit algorithm approach being developed closely mimics the IWC's RMP in many respects.

5.3.4 Comparison of a bycatch limit algorithm approach with PBR

The bycatch limit algorithm (BLA) approach being developed closely mimics the IWC's RMP, particularly in the use of an equivalent of the CLA to set limits to removals. It is currently less well-developed with respect to application to situations where there are multiple management areas.

There are important similarities between a CLA/BLA approach and PBR. The main similarity is that both methods were developed through simulation to ensure robustness to a range of uncertainties and to ensure that application of the final equation(s) allowed defined conservation objectives to be met in the long term (100 years). Both methods use estimates of abundance and take account of uncertainty in those estimates.

The main difference between a CLA/BLA approach and PBR is that the CLA/BLA fits a population dynamics model to a time-series of abundance estimates and removals data but PBR is implemented using a single value of minimum abundance. The CLA/BLA thus estimates relative population level (depletion) and allows implementation of a "protection level" below which limits to removals can be set to zero. This can shorten recovery time to target population levels. If only one estimate of abundance is available to the CLA/BLA, its performance is similar to PBR.

Another difference is that the PBR equation incorporates a recovery factor, the value of which is determined on a case-specific basis. The IWC's CLA is generic for all baleen whales; case-specific aspects are dealt with as part of Implementation Simulation Trials as described above in Section 3.3.3. A BLA could follow this format or species-specific BLAs could be developed.

It is relevant to consider what happens if estimates of abundance or bycatch are not updated. The IWC RMP incorporates a phase-out rule which reduces catch limits to zero over a defined period if a new abundance estimate is not available. The US GAMMS workshop has proposed a similar mechanism for reducing abundance estimates to input to the PBR equation when there is no recent estimate of abundance (Moore and Merrick, 2011). Such phase-out rules could be considered for a BLA approach to reduce safe bycatch limits if new data do not become available in timely fashion.

5.4 Conclusions and recommendations

A bycatch limit algorithm approach makes more use of the available data and allows better scientific advice and should therefore continue to be developed. The implications for implementation of this approach are that time-series of estimates of abundance and removals (bycatch) are required for each Management Area whenever the procedure is updated (for example, every six years-the reporting period for the Habitats Directive and Marine Strategy Framework Directive). For the abundance of most species of cetacean, this would require SCANS-type surveys to occur every six years so that there is a new abundance estimate available for updating. More frequent data collection will occur for coastal bottlenose dolphins. For grey seals and harbour seals the data would come from regular updates of the seal database (see Section 7, ToR E). WGBYC would need to provide comprehensive estimates of annual bycatch for each Management Area at least every six years.

As part of the reforms to the Common Fisheries Policy and the Data Collection Framework, the European Commission requested that ICES provide advice on the use of management frameworks and other mechanisms for determining safe bycatch limits in 2013. The ICES response noted that further work in this area would be required and that: '*This could be in the form of a workshop for invited participants representing managers, scientists and stakeholders. As stressed in the advice, input from management and from the "societal" side is crucial for such a process. We would envisage attendees from relevant parts of the European Commission (at least DG Mare and DG Environment), Member State fisheries authorities, the RACs, relevant intergovernmental bodies (Regional Seas Commissions, ASCOBANS and ACCOBAMS) and relevant NGOs. Whether the workshop should be arranged in Brussels, in ICES or elsewhere depends on an assessment of how to ensure the best input from the policy/management and societal sides.*

Before the meeting it is essential to ask scientists to prepare presentations of models and some model scenarios which can be used during the workshop. Dependent on the timing of the workshop such input could be prepared by ICES expert groups (i.e. WGBYC and WGMME) leading to a workshop in late summer 2014 or alternatively by a smaller invited expert group if the Commission wants the workshop in autumn 2013. We are aware of some relevant preparatory work being done under funding from UK and from ASCOBANS that will be ready in August 2013 for consideration by an ASCOBANS meeting and aims to complete by the end of September 2013.'

The European Commission have yet to respond to ICES regarding this offer.

If a bycatch limit approach to setting safe limits to removals is taken forward, it is pertinent to consider when it may first be able to be implemented. The most recent comprehensive estimates of abundance for all cetacean species except coastal bottlenose dolphins are from 2005/2007 and it would be most appropriate to wait until new data are collected under SCANS-III in 2016, if it takes place. Estimates of abundance from SCANS-III could be available in 2017. Estimates of bycatch from WGBYC would also need to be available at this time. A bycatch limit algorithm approach to setting safe bycatch limits should thus be able to be implemented in 2017. This time scale fits well with reporting requirements under the MFSD.

Given that the lack of agreed conservation objectives is the primary reason stopping further development, WGMME **recommends** that European Commission give serious consideration to ICES offer to host a workshop, with the objective of reviewing different mechanisms for determining safe bycatch limits and finalising conservation objectives for a bycatch limit approach that would enable conservation aspiration to be met.

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6 ToR D Assess the Joint Cetacean Protocol outputs with a view to their contribution to international transboundary reporting requirements (e.g. for Article 17 of the Habitats Directive) and the operationalization of MSFD indicators, targets and appropriate baselines. Consideration should also be given to other approaches, such as those of the Atlantic marine Assessment programme (AMAPPS) which coordinates data collection and analysis for marine mammals and reptiles for population assessments

The first part of this ToR, 'to assess the Joint Cetacean Protocol outputs with a view to their contribution to international transboundary reporting requirements (e.g. for Article 17 of the Habitats Directive) and the operationalization of MSFD indicators, targets and appropriate baselines' was postponed from the 2013 meeting due to delays in the project. Publication was expected in the latter part of 2013 but, unfortunately, this has not occurred. Consequently, it is not possible for WGMME to complete the task at this time, so it has been postponed until 2015. However, in the absence of the JCP, the reporting requirements and operationalization of MSFD indicators and targets for cetaceans is considered in ToR H, the special request from OSPAR (see Section 11).

Whilst the JCP was developed to bring together a large array of disparate data collected for many different reasons and in a variety of different ways, in contrast, in North America, a comprehensive research programme has been adopted in order to assess trends in cetacean and other protected species as required by legislation (i.e. National Environmental Policy Act [NEPA], Marine Mammal Protection Act [MMPA], Migratory Bird Treaty Act [MBTA], and Endangered Species Act [ESA]).

The Atlantic Marine Assessment Program for Protected Species (AMAPPS) is a comprehensive program to assess the abundance and spatial distribution of marine mammals, sea turtles, and seabirds in US waters of the western North Atlantic Ocean. This program includes data collection on dedicated seasonal vessel and aerial surveys, as well as data from tagging and other projects. AMAPPS is the mechanism through which data collection and analysis of the NMFS Northeast and Southeast Fisheries Science Centers and the US Fish and Wildlife Service Division of Migratory Birds is coordinated, with a view to accomplishing six primary objectives:

1) Collect broad-scale data over multiple years on the seasonal distribution and abundance of marine mammals (cetaceans and pinnipeds), marine turtles, and seabirds using direct aerial and shipboard surveys of the US Atlantic waters within the US EEZ and within the Canadian waters in the lower Bay of Fundy and Gulf of Maine (e.g. Figure 6.1);
2) Collect similar data at finer scales at several (~3) sites of particular interest to NOAA partners using visual and acoustic survey techniques;
3) Conduct tag telemetry studies within surveyed regions of marine turtles, pinnipeds and seabirds to develop corrections for availability bias in the abundance survey data and collect additional data on habitat use and life history, residence time, and frequency of use;

4) Explore alternative platforms and technologies to improve population assessment studies;

5) Assess the population size of surveyed species at regional scales; and

6) Develop models and associated tools to translate these survey data into seasonal, spatially explicit density estimates incorporating habitat characteristics.

These objectives are expected to provide data to managers in order to support conservation initiatives mandated under the relevant legislation. The AMAPPS data will be used to support environmental assessments associated with Bureau of Ocean Energy Management (BOEM) and US Navy activities, including anticipated offshore energy projects. These data are being used to improve the assessment of marine mammal stocks as required under the Marine Mammal Protection Act (MMPA); for example, provide data to support updated abundance estimates for US Atlantic oceanic stocks of marine mammals (e.g. Waring *et al*., 2013). In addition, these data are also being used to support programs that monitor the risk of extinction and recovery of the species detected during the surveys, including those species not already covered under the MMPA.

Describing the relationships between the current patterns of density and distribution of marine mammals and seabirds as related to their physical and biological environment enables an understanding of how environmental habitat characteristics drive/control the distribution and density of these animals, as well as a) how to forecast animal density maps to a future time when environmental conditions may have changed, and b) how to discriminate between changes in cetacean populations due to natural environmental variability and changes due to anthropogenic impacts.

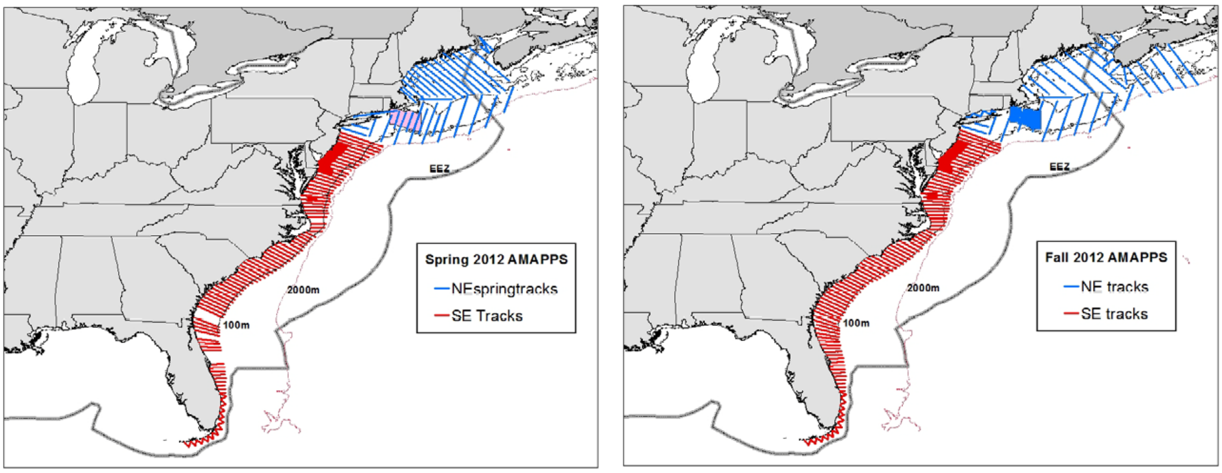

Figure 6.1. Tracklines completed during spring (March–May; A) and autumn (September–November; B) 2012 AMAPPS aerial surveys (taken from NOAA, 2012).

Such an approach (i.e. dedicated large-scale surveys on a regular basis) has been identified as the ideal approach for monitoring these wide spread and highly mobile species when to goal is to obtain accurate population abundance assessments (Forney, 2000; Evans and Hammond, 2003). The AMAPPS approach is somewhat similar to the SCANS, CODA and TNASS surveys (e.g. Hammond *et al*., 2002, 2013; Lawson and Gosselin, 2009; Kaschner *et al*., 2012) that have been undertaken in the Eastern North Atlantic, although these largely European surveys have not been undertaken on such a frequent basis.

The Joint Cetacean Protocol provides an example of a mechanism in which data from these large-scale population surveys can be combined with more localised work from a disparate array of sources, e.g. various governmental organizations, educational organisations, private sector companies and non-governmental organizations. Whilst the Joint Cetacean Protocol data resource is a very large and rich dataset, the effort was very uneven, with large areas of the study region receiving little or no effort, particularly in some seasons and years. Further, almost all data sources came from restricted regions of space or time, in many cases with little overlap between sources. Given the diverse nature of the input data, and the patchiness of the spatio-temporal coverage, modelling of the data will be required to make inferences in trends in distribution and relative abundance over time. Such results are vulnerable to failure of model assumptions, model mis-specification, and other issues. Therefore, for robust inferences about population abundance and trend, it is essential that the platform-of-opportunity data component of the JCP be complemented by periodic, large-scale dedicated surveys that are designed to produce reliable snapshots of abundance (i.e. absolute abundance estimates) at the desired spatial scale. Such surveys can be used as 'anchor points' for the modelling of data contained within the JCP.

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7 ToR E Update on development of database for seals and status of intersessional work, contribution to the operationalization of MSFD indicators, targets and appropriate baselines. Consideration should also be given to other approaches, such as those of the Atlantic Marine Assessment programme (AMAPPS)

7.1 Requirement

To collate information from seal population monitoring programmes across the ICES area and to populate a database so details for different areas can be more easily compared. Furthermore, the database should allow assessments of fulfilment of MSFD indicator assessments regarding temporal trends of abundance and pup production for the management units covered by the database.

7.2 Area of relevance

The area of relevance to ICES is the North Atlantic, including the North and Baltic Seas, where harbour (common) seal, *Phoca vitulina vitulina*, and the Atlantic and Baltic grey seals, *Halichoerus grypus grypus* and *H. g. macrorhynchus*, respectively are found. In the original proposals brought forward by WGMME (ICES, 2008), the countries anticipated to participate included Norway, Sweden, Denmark, Finland, Estonia, Russia, Germany, the Netherlands, Belgium, France, UK, Ireland. It was also thought that, in future, the area covered might be extend to include the Faroe Islands, the Barents Sea (Russia) and the Northwest Atlantic (Iceland, Greenland, Canada and the USA).

To date, the ICES seal database has been developed with the inclusion of data from Denmark, Germany, the Netherlands, Belgium and the UK. Norway, Sweden, Belgium, France and Ireland have also agreed in principle to provide data. Currently, the data within the database are not sufficient for assessment of the proposed core MSFD indicators under Descriptor 1.

In the USA, there has been a long-term commitment to the collection of seal data from a diverse range of sources. Most recently this was formalised through the AMAPPS programme (see Section 6, ToR D). An Oracle database has been developed to hold the data and contains information on sightings and effort data from aerial and shipboard observers, vessel-collected oceanographic and biological sampling data, as well as telemetry data. In addition, these are being linked with satellite-derived and terrain data at appropriate scales for use in density modelling and predictions.

Through the auspices of HELCOM, development of separate databases for seals in the Baltic Sea area is already begun. Given that the seal indicator under descriptor 1 in this area are very similar to those of the OSPAR area (e.g. see HELCOM, 2012), it was felt that a comparison of the current ICES database with that developed by HELCOM would be a useful starting point in the further development work required by this ToR. Additionally, the original intention of the seal database to be a repository for data across the entire ICES is felt to be too ambitious at this time. It was therefore proposed that initially the seal database should be revised such that it meets the needs of MSFD reporting requirements for the OSPAR region.

7.3 Issues

Most importantly, the relevance and longevity of this seal database is entirely dependent on the frequency and extent to which it is populated with information from different countries. The ICES seal database has already experienced problems along these lines, with the ToR on its development and/or updating often being postponed on several occasions (e.g. ICES, 2009; 2011; 2012).

Most organizations that monitor seal populations are very understandably 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 other parties without the consent and knowledge of the contributors. However, European legislation (e.g. Directive 2003/4/EC of the European Parliament and of the Council of 28 January 2003 on public access to environmental information as noted in Article 19 of the Marine Strategy Framework Directive) places a requirement on Member States to make sure all information is publically available. It is unclear at this time whether that requirement relates to raw or processed data.

There is no standard survey methodology in use across all areas or for either species, although there are similarities. Most surveys are carried out from either aircraft or helicopter, for instance, but ground and boat based surveys occur, where aerial survey are not practical. Different components of the local populations of each species may be monitored in different areas. There is variation in survey frequency in different countries. Survey frequency and intensity varies according to the degree of importance of either species in each country, the extent of coastline inhabited by seals and the complexity of that coastline and the substratum on which seals are normally found.

There is also variation in reporting the results of surveys. For instance, harbour seal surveys are carried out either during their summer breeding season or some weeks later, during their annual moult. Data from these surveys may be reported in different ways, as means or trimmed means of several replicate surveys, or as maximum numbers obtained during a season. For analysis purposes it will be desirable to include data from each of the replicate surveys of each season, as this will increase the power of analyses. The Trilateral Group, that collates the results of surveys in the Wadden Sea, 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 separately. Grey seal data are either reported as outlined above for the harbour seals, e.g. in the Baltic and Wadden Sea. The more common approach, however, is to use surveys of annual pup production to model population abundance. This is done in the UK, the USA, Canada and Norway. To allow analysis of these data, the database should include CVs of the modelled abundances.

7.4 Database structure

To date, the current ICES seal population database format is a simple MS Excel workbook. The database will be retained and updated by the ICES database manager. There will be separate worksheets for the following:

- Harbour seal metadata;
- Harbour seal moult surveys;
- Harbour seal pup surveys;
- Harbour seal breeding surveys;

- Grey seal metadata;
- Grey seal pups surveys;
- Grey seal moult surveys;

7.4.1 Harbour seal metadata

Virtually identical with grey seal metadata. 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. 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. Window of opportunity over which surveys are carried out; for both breeding season and moult. If summarised data are reported, the methods for summarising should be reported as well. If a correction factor is used to correct for seals at sea during the surveys, this should be given along with a reference.

7.4.2 Harbour seal moult surveys

This contains the dates and results of surveys carried out during the harbour seal annual moult.

7.4.3 Harbour seal breeding surveys–pups

As above, but reporting numbers of pups counted during surveys. Includes information on whether the data represent pup counts, or whether the counts are converted into an estimate of pup production. If an estimate is produced the CV of the estimate should be reported.

7.4.4 Harbour seal breeding surveys–adults

Dates of surveys and 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.

7.4.5 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 and the method used to do this.

7.4.6 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. If modelled population estimates are reported, CVs should be reported with the estimates.

7.4.7 Grey seal moult surveys

Some countries also monitor grey seal numbers during their moult between December and April e.g. in the Baltic, by the Wadden Sea Trilateral group (regular surveys) and the Republic of Ireland (one survey).

7.5 Recommendation

WGMME **strongly recommends** that ICES members of the OSPAR region provide data so that the seal database be maintained and updated regularly. Such development is considered essential for future MSFD assessments of the OSPAR core set of indicators for seals.

7.6 References

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8 ToR F Outline and review approaches to marine mammal survey design used during pre-consent data gathering and post-consent monitoring in the offshore marine renewables (wind, wave, tide) industry, and provide recommendations for best practice

8.1 Introduction

Of all the environmental investigations undertaken associated with the deployment of marine renewable energy projects, those on marine mammals and seabirds are the most costly. This is principally due to a) the level of regulation applied to these species which places high demand for evidence on impacts during consenting processes and b) the inherent variability of the occurrence of mobile species, for which baseline ecological information (such as population variability) is often limited (WGMME, 2013).

A distinction is often made between pre-consent surveys (including site characterisation studies) and post-consent monitoring of actual development impacts (WGMME, 2013). Pre-consent surveys are usually necessary as a basis for predicting the extent and likely significance of construction impacts. Data requirements for both processes often differ and consistency of data collection efforts before and after consenting can be limited as a result (WGMME, 2013). Currently, data gathering by industry is not always consistent or comparable between development areas and/or jurisdictions, resulting in data gaps and other problems that preclude in-depth impact assessments and ultimately an application of cumulative impacts. There is therefore a need to specify data requirements for assessing impacts of marine renewable energy developments on marine mammals.

This document provides an overview of relevant drivers for data gathering and post-consent monitoring (e.g. policy, legislation) as well as examples of existing practice. This is followed by a perspective on information needs and whether existing data collection efforts are appropriate, followed by a review of methods that could be used to provide such data. We close with two recommendations to ICES.

8.2 Drivers

The marine renewables industry is new, and presents several novel impacts to marine mammals (e.g. collision with tidal turbines, acoustic disturbance through widespread pile driving associated with wind farm construction, etc.). Even where impacts may be similar to those presented by other marine industries, their scale and location is likely to be different. There is therefore no large evidence base from which to inform decision-making, and due to the consequent uncertainty in the prediction of impacts there are difficulties facing decision-makers in consenting renewable energy projects, which necessitates a precautionary approach. Furthermore, many of the regions earmarked for development are areas where understanding of the underlying marine mammal abundance and population dynamics remains limited, presenting further challenges in understanding potential impacts.

Within the EU, the key piece of legislation driving the requirement for better understanding of impacts of marine renewables on marine mammals is the Habitats Directive (Council Directive 92/43/EEC). The EIA Directive (Council Directive 85/337/EEC) requires assessment of impacts to all species and habitats which needs to be presented in the form of an Environmental Impact Assessment (EIA) prior to consent. All cetaceans are European Protected Species (listed on Annex IV of the Habitats Directive) which accords protection to individuals from particular risk, and there is a requirement for Special Areas of Conservation to be designated for seal species and for harbour porpoises and bottlenose dolphins. Fundamentally the protection of these species through the transposition of the Habitats Directive at Member State level aims to achieve Favourable Conservation Status (FCS), which require understanding of the population consequences of impacts. This relates, both to the population as a whole and subsets associated with protected areas (through 'conservation objectives' detailed for the particular site. Impacts which cause changes in mortality and fecundity must therefore be considered in population contexts. The legislation also includes specific clauses considering the effects of disturbance on individuals and the consequence of this for populations.

Outside the EU, similar legislation exists to protect marine mammals and their populations. The principal example is the US Marine Mammal Protection Act (MMPA).

8.2.1 Pre-consent

To enable consent to be granted, developers must provide sufficient evidence, to a suitable standard, within their EIA to support conclusions on the potential impacts that may arise from a proposed development, and the numbers of animals that might be affected. This evidence should be quantitative as far as possible, although subjective interpretation and expert judgement is often also required to determine the 'acceptability' of a level of risk, relative to the conservation objectives. The data required to fulfil this are usually defined on a case by case basis by regulators, to account for the scale of development and the anticipated impacts to species.

To establish the 'significance' of potential impacts, predictions are presented both relative to the population (or other relevant management unit as for seals in Scotland[1]) and within the context of any relevant protected sites.

8.2.2 Post-consent

Once a development is consented, further data collection is required through monitoring of the actual impacts associated with the development. Post consent monitoring undertaken by developers focuses on impacts that were raised during the consenting process, and is also undertaken to validate predictions made within the EIA. Such monitoring effort will allow developers to have reduced uncertainty in future projects, and improves understanding for the wider renewable energy community. This is essential to rationalising the costs of EIA and consenting which would help the industry become more financially viable.

Where levels of uncertainty exist at consenting, then phased development consents have been suggested whereby developers are required to demonstrate their level of impact before progressing to the next stage of their development. This adaptive management approach places particularly high demand on the scientific rigour and robustness of monitoring studies on which further development is contingent.

For regulators and government conservation bodies, areas of investigation should be more strategic to allow broad scale understanding of distributions of animals and

[1] http://www.scotland.gov.uk/Topics/marine/Licensing/SealLicensing

how they are affected by impacts from developments. Results from such work are unlikely to be relevant to any specific development in isolation, but would be of benefit to the industry as a whole as well as to other marine interest groups (WGMME, 2013).

8.3 Existing framework (a Scottish perspective)

Surveys to inform the environmental assessments of offshore wind, wave and tidal-stream developments have taken a variety of approaches. For site characterisation in Scottish waters, a developer, in consultation with environmental consultants, will typically formulate a plan of desk-based and at-sea research to address those environmental drivers identified as significant for consenting. This plan will be presented to the regulator and their statutory advisors for their perspectives on whether these approaches are likely to provide appropriate evidence to inform an eventual consenting decision. These discussions may go through several iterations before survey work starts. Thus the details and extent of environmental investigations required by the regulator will often differ between different developments, depending upon environmental sensitivities, and the likely potential scales of impact.

Once the approach for a particular development has been agreed, surveyors, platforms and analysts will be appointed and surveys begun. For marine mammals in Scottish waters, developers are typically asked to collate and review existing information about their site of interest. Unless sufficient information already exists, developers are often asked to perform monthly boat-, plane- or shore-based sighting surveys year-round for two years, although survey duration requirements depend critically on the individual nature of each development. These surveys are typically (but not always) combined with seabird observations, and may also involve gathering passive acoustic data. Local pinnipeds may also be tagged to investigate their movements and behaviour. Other data (such as haul-out counts of seals or photo-ID of whales or dolphins) may also be required if deemed appropriate by the regulator. The area of interest typically contains the actual development site plus a buffer (usually <10 kilometres wide) around the site perimeter. Surveys of control or comparison sites are rarely requested although regulators do typically require some assessment of the significance of locally derived observations in a broader context.

In addition to individual development-specific survey work, a number of more strategic surveys have been conducted (e.g. The Crown Estate Pentland Firth – Orkney Waters enabling action http://www.thecrownestate.co.uk/energy-infrastructure/wave-and-tidal/pentland-firth-and-orkney-waters/enabling-actions/) to map habitats and investigate broad-scale marine mammal distribution across many developments. A significant example in Scotland has been the strategic aerial sightings surveys for birds with marine mammal observations coming as a by-product (APEM, 2013). Approaches by other ICES Members may be similar or differ.

Once consent has been granted, a development may go ahead with or without specific terms for that consent, which can result in further data gathering requirements. For example, potentially noisy construction activities may require additional monitoring efforts to determine, for example, whether the resulting noise outputs remain within acceptable limits, with mitigation measures to be implemented if those limits are exceeded (BSH, 2007). In other cases, particularly those involving novel energy-extraction technologies, construction may be staggered based on the results of specific impact research activities (www.scotland.gov.uk/Topics/marine/Licensing/marine/scoping/MeyGen).

8.4 Information needs

As indicated above, data requirements for marine renewable energy developments can vary widely, depending on the development scale, number of species involved, and legislative requirements. There are, however, several basic pieces of information that regulators require, both during the consenting process and as part of a post-consenting monitoring strategy. These include the following:

- Species presence;
- Regularly updated abundance estimates for all management units (populations) for all species of interest;
- Trend analysis of abundance estimates over time and information on demographic parameters (e.g. reproductive rates, survival, etc.);
- Assessments of temporal variability (e.g. seasonal cycles).
- Detailed information on habitat usage, including reproductive activity, foraging, migratory pathways, local residency, etc.
- Connectivity between development sites and protected areas;
- Local environmental data to aid in habitat modelling.

Such data may be sufficient for many developments (e.g. offshore wind). For tidal energy development sites, in particular, the following additional information will also be required:

- High-resolution marine mammal survey data collection across tidal cycles, across seasons, using a range of methodologies as needed;
- Three-dimensional distribution of animals in the water column.

The most basic information required for any development is an assessment of which marine mammal species occur in a potential development site. This can be obtained through visual and/or acoustic survey effort although multiple visits will be required to assess the presence of less common and/or migratory species.

Moving beyond animal presence, animal abundance (or density) is a far more informative parameter to assess the significance of sites to marine mammals, both in terms of absolute numbers on the site and as a percentage of populations whose range encompasses the proposed development site. Without reliable abundance estimates, it becomes very difficult to put any potential impact resulting from development activities into a broader context. Moreover, regulatory drivers for species conservation typically require an assessment of impacts on populations or management units.

The scale on which surveys are undertaken is important. To date, much survey effort in European waters has occurred at the level of individual development sites. This spatial scale is typically inappropriate to undertaking surveys of large, highly mobile marine species that may be migratory. Moreover, some impacts (e.g. those related to acoustic disturbance), will affect animals many tens of kilometres beyond the development site. Comprehensive, synoptic, strategic survey efforts across large areas (cf. Hammond *et al.*, 2002, 2013; Reeb, 2013) are more appropriate. However, large-scale surveys may not capture potentially significant variability of distribution across comparatively small spatial and temporal scales. This is particularly relevant to tidal and wave energy developments which occur in sites that are both highly variable and potentially quite distinct from surrounding marine environments. There is therefore a need to incorporate small-scale high-intensity surveys (undertaken by individual developers) amongst a larger framework of broad-scale strategic surveys (undertaken by regulators at a regional or international scale; WGMME, 2013). Some areas (e.g. breeding or foraging areas, haul-out sites for seals, etc.) may be of considerable significance to marine mammal species, and may require additional monitoring effort.

In addition to establishing abundance, it is important to establish how animals use the habitat at development sites. While it is usually difficult to assess behaviour of marine mammals underwater, some observations (e.g. visual observations of foraging, stomach contents of stranded or bycaught individuals, etc.) can provide general insights into the importance of particular sites for marine mammals that make use of them. Three-dimensional diving profiles of marine mammals provide important information on their usage of the entire water column (including foraging activity) and can be informative in subsequent modelling efforts to assess risks, e.g. collision risks involving tidal turbines. Animals' likely responses to sound associated with marine renewable energy devices (such as operational tidal turbines) are similarly important for risk assessment.

There are important differences between developments impacting a local population of animals resident in a particular area vs. a succession of naïve animals migrating through an area. Residency time, movements and the average amount of time individual animals spend in the proposed development area, can therefore be very informative.

In a similar vein, it is important to consider the spatial scale across which animals may be impacted by a development. Disturbance of animals due to, for example, acoustic impacts may extend over a considerable distance beyond the nominal spatial footprint of the proposed development; adequately assessing large-scale spatial redistribution around such developments may require considerable monitoring effort and may be difficult to detect using traditional methods (but see Dähne *et al.*, 2013 for a successful monitoring strategy). There is, however, a need to determine across what distances animals might be displaced, and the extent of recovery once the impacting activity (e.g. pile driving) is reduced or eliminated.

A significant problem in assessing impacts of marine renewable energy on marine mammals lies in linking impacts on individual animals to population-level consequences. Assessing such broad-scale impacts requires assessment of long-term effects on animal energetics, health, reproductive rates and survival. Collecting such data on populations of long-lived, wide-ranging marine mammals is likely to require detailed long-term monitoring programs. The recent development of PCOD (Population Consequences of Disturbance) modelling methods provide an example of a theoretical approach where a wide range of data are used to assess potential population consequences (Lusseau *et al.*, 2012).

With many marine renewable sites (especially offshore wind) being considered for concurrent development, the potential cumulative impacts of multiple developments need to be considered carefully. At present there is little information on how multiple environmental pressures might collectively impact marine mammals. Moreover, development of offshore marine renewable energy sites co-occurs with existing anthropogenic pressures including shipping, fisheries, offshore oil/gas exploitation and others. The cumulative effects of these disparate industries on the long-term conservation status of marine mammal populations are only just beginning to be explored.

8.4.1 Tidal and wave sites

Tidal and wave sites pose their own special problems for monitoring. They are often small sites with physical and biological conditions that are quite different from surrounding waters meaning that there is limited justification for extrapolating from the larger areas outside them, except to place any changes due to development impacts in the context of the wider population. Tidal sites in particular require directed and dedicated surveys. The strong currents running at tidal sites make them challenging locations in which to survey, as the turbulent waters in tidal rapids lead to poor sea states that compromise sightings surveys. Water currents which are quite large in relation to vessel speed raise fundamental issues about the frame of reference to use when designing surveys, because the volume of water surveyed can be very much greater than would be suggested by calculating the distance of line transect covered (over the ground). In many locations it has been shown that the distribution of animals varies enormously throughout the tidal cycle. For example, animals are found in different parts of tidal rapid system during ebb and flood tides (Wilson *et al*., 2013). Thus, it is important for surveys to cover all areas at all states of the tide. Times when current are running most strongly are particularly relevant because this is when tidal turbines will be turning most rapidly.

Wave sites pose similar problems in that data on marine mammal distribution, abundance, etc. need to be gathered across the range of wave conditions likely to be encountered at the site, rather than solely during periods of calm weather. Strong wave action may drive differences in animal distribution, dive cycles and surfacing behaviour, and may also make devices more difficult to detect through increased ambient noise levels. Clarifying distribution, abundance and habitat use in such environments will almost certainly require an innovative monitoring approach in which different complementary techniques are used.

8.5 Methods

As discussed above, to be able to consent a development regulators usually have to put the envisaged impacts (e.g. number of animals disturbed or subjected to hearing damage) into a population context. To be able to do this they will typically need an abundance estimate and associated confidence intervals. Various methods exist to fulfil the information requirements outlined above. Abundance estimation of marine mammals is a well-developed field with a well-established methodology and literature (e.g. Buckland *et al*., 2001, 2004; Thomas *et al*., 2010). Typical approaches involve some form of line-transect survey analysed using DISTANCE density estimation techniques (Thomas *et al*., 2010). Robust absolute abundance estimation may, however, require considerable data collection efforts to obtain sufficient numbers of detections (Buckland *et al*., 2001).

8.5.1 Densities

There are a number of approaches that can be used to calculate absolute densities. The methods chosen must be capable of providing the required information (absolute abundance, reliable index of abundance, etc.) in conjunction with confidence intervals to illustrate the precision of the estimate. The techniques should be open and transparent and be generally accepted by the research community as being reliable and fit for purpose.

Methods that a consultant might propose to use will likely vary from survey to survey. Relevant considerations include location of the site, species of interest, expected weather conditions, length of daylight and the potential for combining marine mammal monitoring with ornithological or other surveys. Although there have been a number of attempts to assess the relative merits of different approaches (e.g. Hammond *et al*., 2002; MacLeod *et al*., 2011; Thompson *et al*., 2014), the conclusions of these have not always coincided and these assessment should generally be made on a case by case basis following the approaches outlined in these reports.

The precision of abundance estimates is an important consideration. Some estimates are likely to be far less precise (i.e. possess far wider confidence intervals) than others. Various factors may influence estimate this including varying detection rates (affected by animal density, animals being inconspicuous, shy or otherwise difficult to detect, poor surveying conditions [e.g. high sea states] and survey efficiency; see also Buckland *et al*., 2001). Precision is typically expressed as Coefficient of Variation, or CV. Usually a developer should choose an approach that promises to provide the most accurate density estimates over the required area with the tightest confidence intervals for the lowest cost (Thompson *et al*., 2014). If regulators clarify that they intend to use precautionary values of densities in their assessments, based for example on some predetermined upper confidence interval, then a developer will be rewarded for conducting better surveys and will also be incentivised to spend more on surveys in more sensitive areas where precise density estimates are more critical.

Generally, low CVs can be achieved by carrying out high-quality surveys, collecting accurate and consistent field data, allowing for as many sources of variation as possible, by concurrently collecting and incorporating environmental covariates into analyses and, finally, by increasing the amount of survey effort. Thus, there are likely to be trade-offs between survey quality and quantity, and the value of collecting good quality field data should be obvious. More generally there is a need to consider both large-scale synoptic survey effort across large areas and small-scale focused effort in areas of particular development interest (e.g. the US Bureau of Offshore Energy Management's AMAPPS program; Reeb, 2013).

8.5.2 Boat-based visual surveys

The most widely applied technique for assessing the densities of marine mammals at sea has been for a team of experienced observers to conduct visual surveys from vessels. All marine mammals, including seals, can be monitored using this technique (assuming various theoretical assumptions are met [Buckland *et al*., 2001], such as the assumption that the detection probability of animals on the trackline is 100%; this is typically expressed as g(0)=1). When appropriate line-transect techniques are applied absolute density estimates can be generated for most if not all species. Data on other parameters of interest such as oceanographic parameters, presence of seabirds, shipping activity, fishing gear density etc. can be collected from the same vessel. Because boats move relatively slowly and powerful binoculars can be used, sightings can be scrutinised to facilitate more reliable species identification and group size estimation. It is also possible to conduct surveys with a "closing mode" design, where the survey can be temporarily suspended and the vessel can close with sightings of interest allowing species of interest to be scrutinised in more detail (although care must be taken to prevent such a design from yielding biased density estimates; Dawson *et al*., 2008).

In many surveys of wind farm sites undertaken in UK waters to date, however, techniques to provide absolute abundance have not been applied. Often marine mammal surveys have been adjuncts to seabird surveys. Seabird surveyors collect data in a manner which is incompatible with quantitative marine mammal surveys (Camphuysen *et al*., 2004) and it is difficult for a single observer team to collect both sets of data at the same time. One obvious solution would be to have two complete teams of both seabird and marine mammal observers on the same vessel at the same time. This has rarely been done, in part because of the difficulty of accommodating two complete observer teams on the same vessel. An additional concern with conducting both surveys at the same time from the same vessel is that the marine mammal surveys will usually need to cover a much larger area than seabird surveys, reflecting the much greater range at which impacts such as underwater noise affect marine mammals. The need to base survey requirements on the least detectable species of interest (typically marine mammals rather than seabirds) has been identified previously (e.g. MacLeod *et al*., 2011; WGMME, 2013). Varying levels of observer expertise are also cause for concern among regulators. Therefore marine mammal observers should possess sufficient practical experience to survey effectively.

8.5.3 Boat-based towed hydrophone surveys

Towed hydrophones can provide data on the occurrence of marine mammals that vocalise consistently. Advantages of Passive Acoustic Monitoring (PAM) compared to visual detection include the fact that it is less affected by weather conditions so that monitoring can continue at higher sea states, under poor visibility and overnight. Data collection and analysis are highly automated, requiring smaller field teams and resulting in more consistent data collection which can be reviewed and reanalysed ashore. However, not all species vocalise so it is certainly not a solution for all marine mammals. At European wind farm sites PAM has been particularly useful for assessing densities of harbour porpoises. For porpoises, ranges from the trackline can be calculated using target motion analysis when simple stereo arrays are utilised. These range data allow detection functions to be calculated facilitating quantitative distance based line transect analysis.

Unless a survey was narrowly focused on an amenable species, such as harbour porpoise or sperm whales, PAM would normally be used in conjunction with boat-based visual survey effort rather than in place of it. In such cases an additional advantage of PAM is that it can provide an independent mode of detection allowing mark-recapture methods to be used to determine g(0) for both visual and acoustic surveys, which in turn allows absolute abundance to be calculated (Leaper and Gordon, 2012). Some of the few wind farm pre-consent surveys that have provided absolute abundance estimates to date have utilised joint visual acoustic methods to calculate g(0) (Brookes *et.al*., 2013).

8.5.4 Aerial visual surveys

Aerial visual survey is another standard technique, and methods for analysing data from aerial surveys using Distance techniques are well developed. G(0) can be calculated using tandem surveys or "racetrack" techniques (Hammond *et al*., 2013) and often the value of g(0) measured on similar surveys in similar areas is used to inform absolute abundance calculations for surveys where data to inform g(0) cannot be collected. Although the hourly costs of aerial surveys are high they can cover larger areas in a relatively short period of time. The relative costs of carrying out aerial surveys is highly dependent on the availability of suitable aircraft, the distance of the survey block from the aircraft's base and the expected weather conditions on the site (Thompson *et al*., 2014). Given its speed of travel and restricted size, it will be more difficult to collect data from all marine mammal types from an aircraft (e.g. those species that dive deeply/for extended periods) and difficult to combine marine mammal and bird survey teams on the same platform. Ancillary environmental data also cannot be collected by this method.

Relatively new developments involve the use of either still or high definition video images taken from planes. Usually, surveys are being flown for bird monitoring and the same images are also examined for marine mammals. Not all methods have been published or openly peer reviewed as some firms consider details regarding camera models, settings used and methods of extracting data from images to be confidential. It is therefore difficult to assess their suitability for marine mammal surveys. A positive aspect is that they provide auditable data in the form of still images or video clips that can be reviewed after the surveys and assessing the area covered is very straightforward- being the aggregate of the area in each image. However, some shortcomings have been highlighted (Thompson *et al*., 2014). Species identification may be difficult on images, and no truly independent assessment of accuracy has been published. There are also concerns about quantifying the extent to which factors such as sea state and turbidity could affect the detectability of animals. Currently, there is no estimation of availability so an estimate of g(0) cannot be used to allow for calculation of absolute abundance. Perhaps the most fundamental concern is the low number of sightings reported from such surveys (Thompson *et al*., 2014). This is very much to be expected because strip widths covered by the cameras are quite narrow, each section of the strip width is sampled near-instantaneously and marine mammals spend the majority of their time underwater. This low sighting rate means that more track lines will need to be flown to achieve an adequate sample which may obviate the cost advantages of these techniques. Fundamentally, it is difficult to assess the adequacy of methods which are not open and transparent and have not been scrutinised by the scientific community.

8.5.5 Independent passive acoustic detectors

Independent passive acoustic detectors offer a robust method of assessing presence of different vocalising marine mammal species over a wide range of temporal and spatial scales. These devices are typically moored to the seabed where they record or detect animal vocalisations and/or ambient noise levels for considerable periods of time, providing data at high temporal resolution for particular locations. Multiple detectors can be used to assess the approximate location and position of vocalising animals in the water column, and vocalisation data can provide information on species identification, migratory routes, social behaviour, foraging, etc. Effective detection radii around such detectors are likely to vary considerably under changing environmental conditions, complicating attempts to use these data for density estimation (but see Tougaard, 2008). Their use in marine renewable energy development monitoring programmes has to date largely concentrated on offshore wind farm sites (Scheidat *et al*., 2011; Thompson *et al*., 2004). Use of these devices in tidal and wave energy sites requires careful consideration, as the extreme conditions in these sites necessitate increased complexity and cost of mooring construction, deployment and recovery (Wilson *et al*., 2013). Nevertheless, such devices can provide very useful information: Gordon *et al*. (2011) determined densities of harbour porpoises over all states of the tide at two tidal rapid sites off Wales using a range of different survey methodologies. They found that moored static monitoring detectors were the most effective means of revealing temporal patterns. Wilson *et al*. (2013) report on using drifting acoustic detectors for porpoises in tidal environments as a means of sampling large areas across tidal cycles.

Gordon *et al*. (2011) report on preliminary trials of the use of drifting vertical arrays to measure dive depth and underwater behaviour of porpoises in tidal rapids. Extensive work has been conducted since then to develop a more capable and usable system that can monitor three-dimensional tracks of porpoises in tidal areas. This system is likely to work with any animals that vocalise sufficiently frequently (odontocete cetaceans) but may not work as well for baleen whales or seals (Macaulay *et al*., 2013).

8.5.6 Telemetry

Seals are very amenable to being studied with telemetry because individuals can be captured onshore and equipped with telemetry tags glued to their pelts. In areas such as northern European waters, where there is a long history of such studies, telemetry data have been used to assess densities in offshore areas (Jones *et al*., 2013). The approach consists of using telemetry data to develop a model of animal movement and habitat preference. Telemetry can be used to predict where animals from particular haul-out sites are likely to spend their time at sea and these probabilities of occurrence are "scaled" by the size of known haul-outs to arrive at predicted at-sea seal density maps for offshore waters (Jones *et al*., 2013). One assumption is that all haul-out sites are known, which may not always be the case (e.g. for animals that travel great distances to reach a particularly desirable foraging site). It will always be sensible to test the predictions from these models against direct observation data from visual surveys. As development sites become smaller and more specific (tidal or wave sites for example) it will be increasingly difficult to apply this sort of approach. In addition to surface movements, telemetry can also be used to study diving behaviour, which can provide an indication of specific feeding areas.

Collision risk is considered one of the most serious risks for marine mammals in tidal areas and to assess this it is important to measure dive depth and underwater behaviour of marine mammals in these areas. Telemetry devices have been used to record such data for seals (e.g. McConnell *et al*., 2013). It is likely that telemetry could also be used with large whales. However, telemetry is typically not feasible for studying small cetaceans such as porpoises except under very specific circumstances when animals can be captured alive (Johnston *et al*., 2005; Sveegaard *et al*., 2011).

8.5.7 Stranding schemes

A considerable amount of data of potential relevance to marine renewable energy developments can be obtained from stranding schemes and related carcass recovery efforts (including animals bycaught in fishing gear). Stranding records provide generic indicators of species diversity and overall distribution, although post-mortem transport of carcasses needs to be considered. Depending on the state of the carcass, a host of biological samples and demographic data can be collected which can inform broader questions such as reproductive rates, longevity, growth rates, contaminant loads, population structure, genetics and cause of death (e.g. Westgate and Read, 2007; Murphy *et al*., 2009; Slooten and Barlow, 2003; WGMME, 2012). Such data provide relevant context for impact assessments of marine renewable energy developments. For example, work is planned in the UK to assess the kinds of injuries likely to be sustained by marine mammals as a result of colliding with tidal turbine blades (Sparling *et al*., 2013). Such studies are strategic and unlikely to be undertaken by individual developers. In order to assess impacts of tidal energy development robust stranding schemes need to be put in place such that there is at least some probability of detecting the results of collision events, and developers could contribute to such schemes around their particular sites.

8.5.8 Predictive habitat modelling

The distribution of marine mammals is unlikely to be uniform across an area. It is likely that animals distribute themselves to maximise feeding opportunities and minimise risks from predation of exposure to noise. Densities can be predicted by models that incorporate factors such as water depth, bottom type or other known habitat predictors which may themselves be correlated with factors such as foraging conditions (Booth *et al*., 2013; Redfern *et al*., 2006; Mackenzie *et al*., 2013). Such models can both provide insight into how the animals are using the study site and why it is important to them, and also, by accounting for variability of densities, provide more precise density estimates and an improved ability to detect change. For both of these reasons predictive modelling is useful and recommended. The output of such predictive models can be a two-dimensional density surface for the surveyed area. If there is a temporal component to density distributions (such as successive surveys across seasons, tidal cycles, etc.) then multiple density surfaces covering appropriate time periods can be generated. These density surfaces can also be useful in providing a more detailed assessment of the size of impacts that might be expected in different parts of a large study site. While this approach allows interpolation across a surveyed area, extrapolation far outside it is not recommended (unless just outside the surveyed area, with environmental covariates that are within the range of those covered by the surveys). Such models are likely to be most robust when using environmental data from the site itself, particularly when considering energetic environments such as tidal channels. This requires collection of environmental data at the site itself concurrent with, or closely linked to, marine mammal survey efforts.

8.5.9 Photo identification

If animals are sufficiently well marked, and available to be photographed, photo-identification (photo-ID) techniques can be used. Photo-ID studies can be particularly useful to marine renewable energy development assessments in several respects. First, photo-ID studies can reveal the extent of any “connectivity” between animals using a protected area and a potential development site. Second, such studies can provide an estimate of the size of the “population” of animals using a site through mark–recapture analysis. This is most likely to be useful at sites where independent photo-ID studies are already underway. In Europe photo-ID has been useful for bottlenose dolphins, for which there are now time-series of more than twenty years for some populations (e.g. Cheney *et al*., 2013), and there would be scope for applying the approach to other species such as white-beaked dolphins and minke whales.

Photo-ID has also been used to determine demographic parameters such as rates of fecundity and survival in species such as bottlenose dolphins and harbour seals (Cordes and Thompson, 2013), although in the latter case this requires access to a breeding haul-out, close enough to take high quality photographs of individual seal faces to match their pelage. Initial analyses to determine whether rates of fecundity and survival can be estimated are currently underway for UK bottlenose dolphin datasets. Since demographic data require an understanding of individuals within a population, photo-ID studies are likely to be among the few ways in which such data can be obtained. Residency time can also be assessed by means of photo-ID schemes, but may be difficult to establish for all species in all areas, particularly when animals are difficult to identify (e.g. harbour porpoise). Finally, large-scale strategic photo-ID studies provide information on the sizes and trends of entire populations (e.g. humpback whales /Project JoNAH) against which results at marine renewable energy sites can be compared.

8.5.10 Further considerations

Beyond the data requirements outlined above, various processes have to be put in place in order for monitoring strategies to be successful. Once baseline data have been gathered, a gap analysis is essential to identify where information is still lacking and how such gaps might be addressed. This will enable better prediction of impacts, setting of thresholds (of impact at the population/management unit level) to support consenting, and set the basis for design of adequate (post-development) monitoring programmes. A collaborative approach between industry and regulators is recommended to ensure gaps are addressed effectively.

Before undertaking any survey programme, developers should seriously consider the statistical basis underpinning survey designs in order to appreciate the realistic costs of monitoring for impacts (particularly of adaptive management approaches where further development is contingent on results). There needs to be a more rational basis for monitoring requirements that is both cost-effective and a reasonable burden for industry. Considering carefully the spatial and temporal scale required could also lead to a rational basis for collaboration between different developers. Such collaboration between developers across adjacent sites, where it will be difficult to distinguish impacts between projects, should be strongly encouraged. This is relevant to pre-consent prediction of impacts and post-consent monitoring to fulfil licence conditions, and should be regulator-led.

Furthermore, data related to marine renewable energy projects should be collected in such a way as to enable regional understanding of issues such as noise and population consequences. Data sharing, consistency and accessibility are therefore likely to become increasingly important issues in future. In this way, the marine renewable energy sector will become more intimately integrated into broader marine planning processes.

8.6 Recommendations

8.6.1 Top-level recommendation: transparency of monitoring methods

There is a wide range of monitoring methodologies available to assess marine mammals at marine renewable energy development sites, but not all techniques are equally appropriate to all sites. Moreover, assessing the suitability of techniques and the quality of resulting survey data can be hampered by incomplete reporting of methodological details by developers. Commercial sensitivities may further complicate efforts by regulators and others to compare monitoring techniques on their respective merits.

WGMME **recommends** that regulators and policymakers should require the use of open, transparent and reproducible survey and monitoring methodologies to assess potential impacts on marine mammals. Furthermore, for line-transect surveys, the data should be fit to provide absolute densities. For all monitoring, the use of established and peer-reviewed methods is encouraged, acknowledging that new innovations or methods may arise. Methods associated with such new techniques should be sufficiently well described so that conclusions arising from these techniques are reproducible. Data from surveys should be made publicly available in formats that allow future reanalysis (for example using JCP-type protocols).

This top-level recommendation incorporates a more specific recommendation:

8.6.2 Appropriate monitoring coverage

Particularly for wave and tidal sites, there is a need to ensure that monitoring efforts occur across appropriate environmental conditions for that site. This means that survey effort (in its broadest sense) should not be confined to calm weather/low current, but that approaches are needed to ensure data are also gathered during more extreme environmental conditions.

WGMME **recommends** that, in order to characterise sites of interest for marine renewable energy extraction, regulators should require that monitoring take place across the range of significant environmental conditions likely to alter the distribution or behaviour of marine mammals. This particularly applies to wave sites across the normal range of wave conditions and to tidal sites in different flow regimes. Doing so may require combining information gathered using a variety of open, transparent and reproducible monitoring methodologies.

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9 ToR G Special request: Interactions between wild and captive fish stocks (OSPAR 4/2014)

a) Recalling the conclusion of the QSR 2010 that mariculture is a growing activity in the OSPAR maritime area, EIHA 2012 considered the potential for increasing environmental pressure relating to the growth of this industry. As yet this is not an established work stream within EIHA, and Contracting Parties have requested that more information be brought forwards on this issue. This was reiterated by EIHA 2013.

b) Mariculture has a number of associated environmental pressures such as the introduction of non-indigenous species, which can have ecological and genetic impacts on marine environment and especially on wild fish stocks; in addition, pressures from mariculture might include:

   i) introduction of antibiotics and other pharmaceuticals;

   ii) transfer of disease and parasite interactions;

   iii) release of nutrients and organic matters;

   iv) introgression of foreign genes, from both hatchery-reared fish and genetically modified fish and invertebrates, in wild populations;

   v) effects on small cetaceans, such as the bottlenose dolphin, due to their interaction with aquaculture cages.

c) EIHA proposes that OSPAR requests ICES to provide:

   i) an update on the available knowledge of these issues;

   ii) concrete examples of management solutions to mitigate these pressures on the marine environment;

   iii) advise on which pressures have sufficient documentation regarding their impacts to implement relevant monitoring and suggest a way forward to manage these pressures.

d) It may be appropriate to explore cooperation with other competent authorities working in this field, such as the European Food Safety Authority with respect to disease transfer or parasites, or the North Atlantic Salmon Conservation Organisation (NASCO), in particular with respect to existing cooperation between NASCO and ICES on issues pertaining to pressures from mariculture.

For the purposes of the meeting, and with advice from ACOM, it was decided that this request should be summarised as 'Review the effects of aquaculture on marine mammals; where possible, provide examples of management solutions that have mitigated any pressures from aquaculture on marine mammals; recommend which pressures have sufficient documentation regarding their impacts to implement relevant monitoring and suggest ways forward to manage these pressures.'

9.1 Introduction

The main types of marine aquaculture operations include finfish, shellfish and seaweed, of which finfish and shellfish operations are most likely to come into conflict with marine mammals (Tlusty *et al.*, 2001; Würsig and Gailey, 2002). Open aquaculture systems are normally used in marine aquaculture, which means that pens with mesh walls are submerged in natural bodies of water with no solid barriers separating the farm area from the surrounding environment. The open systems rely on natural water currents to replenish oxygen and remove wastes from the facility (Cottee and Petersan, 2009). Finfish farming includes the extensive raising of fish species such as salmon, sea bass and sea bream. The presence of the fish themselves can act as an attractant, for seals in particular. The fish feed that falls through the pens can attract substantial numbers of wild fish to the vicinity of the fish farm, which, in turn, can act as attractant to foraging marine mammals (Würsig and Gailey, 2002; Cottee and Petersan, 2009). Shellfish aquaculture is often associated with extensive farming of oysters, mussels and shrimp and, in contrast to finfish aquaculture, there is little direct conflict with marine mammals. The industry rarely requires nets or cages, but the racks are held in the correct position above the seafloor by buoys and ropes in long parallel lines. In some areas, piling is used to provide a secure foundation for mussel seed collection devices (e.g. Netherlands, Haan and Burggraaf, 2011). As a result, some concerns have been raised regarding the potential impact of entanglement and habitat exclusion on marine mammals (Watson-Capps and Mann, 2005).

9.2 Distribution of aquaculture

Table 9.1 provides a summary of aquaculture in the North Atlantic. It should be noted that the production statistics are taken from the FAO website and include production from contained systems (freshwater as well as marine) unless otherwise stated.

Table 9.1. Summary of main species produced in aquaculture by North Atlantic nation in order of production volumes.

Country	Reported Production in tonnes (year)	Species
Norway	1 300 000 (2010)	Salmon (*Salmo salar*) and rainbow trout (*Oncorhynchus mykiss*) farming dominate coastal production. Interest in Atlantic cod (*Gadus morhua*), the Atlantic halibut (*Hippoglossus hippoglossus*) and the spotted wolffish (*Anarhichas minor*) has been developing. European flat oyster (*Ostrea edulis*) culture has been ongoing for more than a hundred years. The Pacific cupped oyster (*Crassostrea gigas*), mussels (*Mytilus edulis*) and scallops (*Pecten maximus*) have been introduced more recently.
USA	420 000 of which 15 000 occurs in marine pens in the North Atlantic (2012)	81% of production is focused on the freshwater channel catfish (*Ictalurus punctatus).* The primary marine finfish being cultured are the Atlantic salmon (*Salmo salar*) and the white sturgeon (*Acipenser transmontanus*), while others in the early stages of commercialization include sixfinger threadfin (*Polydactylus sexfilis*), cobia (*Rachycentron canadum*), longfin yellowtail (*Seriola rivoliana*) and Atlantic cod (*Gadus morhua*). Shellfish culture in the North Atlantic includes American cupped oysters (*Crassostrea virginica*), hard clams (*Mercenaria mercenaria*), Manila clams (*Venerupis philippinarum*) and blue mussels (*Mytilus edulis*).
Spain	264 000 (2012)	Approximately 90% of production is for marine species, the majority being mussels (*Mytilus galloprovincialis*), which only account for 40% of value. At present, marine fish species cultivated include gilthead sea bream (*Sparus aurata*), turbot (*Psetta maxima*), European sea bass (*Dicentrarchus labrax*), European eel (*Anguilla anguilla*), blackspot sea bream (*Pagellus bogaraveo*), meagre (*Argyrosomus regius*), common sole (*Solea vulgaris*) and tilapia. Shellfish production includes clams (*Ruditapes philippinarum* and *Ruditapes decussatus*). Other species nearing commercial production are the red sea bream or snapper (*Pagrus major)*, the common sea bream (*Pagrus pagrus*), red mullet (*Mullus* spp.), and the octopus (*Octopus vulgaris*).
France	205 000 (2012)	Marine production is dominated by molluscs; mainly oyster (*Ostrea edulis*) and mussels (*Mytilus edulis*). European sea bass (*Dicentrarchus labrax*), gilthead sea bream (*Sparus aurata*) and turbot (*Psetta maxima*) dominate the finfish production, although salmon (*Salmo salar*) is also cultivated. Currently, the developmental potential of red drum (*Sciaenops ocellatus*, also known as channel bass, redfish or spottail bass) and the sturgeon caviar (*Acipenser baerii*, a Siberian sturgeon species) are being investigated.
UK	198 000 (2011) of which 165 000 occurred in Scotland	Mainly salmon (*Salmo salar*), but also halibut (*Hippoglossus hippoglossus*), sea trout (a sea-running form of the Brown trout, *Salmo salar*) and shellfish, particularly mussels (*Mytilus edulis*), and also oysters (*Ostrea edulis*).

Country	Reported Production in tonnes (year)	Species
Canada	174 000 of which 132 000 is finfish in North Atlantic and 41 000 shellfish.	Canadian aquaculture is dominated by the marine sector in which Atlantic salmon (*Salmo salar*) and blue mussel (*Mytilus edulis*) account for 95 percent of production. Finfish culture in the Atlantic is dominated by salmon production. Besides mussels, the American cupped oysters (*Crassostrea virginica*) are grown in the Atlantic provinces. Expansion fo the finfish sector to producing sablefish (*Anoplopoma fimbria*), Pacific halibut (*Hippoglossus stenolepis*) and the wolf-eel (*Anarrhichthys ocellatus*) is ongoing.
Faroe Islands	63 000 (2012)	Fish farming is the second most important contributor to the Faroese economy. Although the industry began with rainbow trout, in 2012, the entire commercial production was of Atlantic salmon (*Salmo salar*). The potential for diversification into other species, in particular cod (*Gadus morhua*), is currently being explored. The large and genetically distinct cod from the Faroe Bank has an especially rapid growth rate.
Netherlands	46 000 (2012)	Mainly blue mussels (*Mytilus edulis) and* oysters (*Ostrea edulis* and *Crassostrea gigas)*. Limited farming for European eel (*Anguilla anguilla*), African catfish (*Clarias gariepinus)* and tilapia (*Oreochromis niloticus*), turbot (*Scophthalmus maximus*), sole (*Solea solea)*, barramundi (*Lates calcarifer*), pike-perch (*Stizostedion lucioperca)* and whiteleg shrimp (*Penaeus vannamei).*
Ireland	46 000 (2010)	85–95% of production is for Atlantic salmon (*Salmo salar*) and trout (*Oncorhynchus mykiss*). Mussels (*Mytilus edulis)*, Pacific oysters (*Crassostrea gigas*), native oysters (*Ostrea edulis*), clams (*Tapes semidecussatus* or *Ruditapes phillipinarum*) and scallops (*Pecten maximus* and *Chlamys opercularis*) are the main shellfish species produced. Commercialisation of cod (*Gadus morhua)* and abalone *(Haliotis tuberculata and Haliotis discus hannai)* are being investigated.
Germany	26 000 (2012) of which approximately half is marine culture.	Industry is dominated by freshwater production of rainbow trout (*Oncorhynchus mykiss*) and common carp (*Cyprinus carpio*). Other freshwater species include pike (*Esox lucius*), zander (*Sander lucioperca*) and tench (*Tinca tinca*). The most important marine species cultured in Germany is the mussel (*Mytilus edulis*). Some marine finfish species include turbot (*Psetta maxima*), European sea bass (*Dicentrarchus labrax*) and macroalgae (e.g. *Laminaria saccharina*), although this is mainly in contained systems.
Denmark	39 000 (2012) of which 10 300 was finfish in marine pens and 2600 shellfish	The main product from offshore cages is large rainbow trout (*Oncorhynchus mykiss).* An essential by-product is the roe, which is salted and marketed as 'caviar'. Small quantities of turbot fry are produced for export for further ongrowing, mainly in Southern Europe; in addition, some plaice are produced for restocking purposes. Mussels (*Mytilus edulis*) and the European flat oyster (*Ostrea edulis*) have been farmed from time to time in small quantities within the Danish fjords. Due to the risk of ice during cold winters, the sea around Denmark is not suitable for mariculture all year around. Cages are taken ashore during winter.

Country	Reported Production in tonnes (year)	Species
Sweden	14 000 (2012)	The industry is dominated by the production of rainbow trout (*Oncorhynchus mykiss).* Other species such as brown trout and salmon (*Salmo salar*), whitefish (*Coregonus albula*) and pike (*Esox lucius)* are also produced. Mussels (*Mytilus edulis)* and crayfish (*Pacifastacus leniusculus*) are also cultivated.
Portugal	8000 (2012)	88% of production is marine species such as Sea bass (*Dicentrarchus labrax*), sea bream (*Sparus aurata)*, oysters (Crassostrea angulata), turbot (*Psetta maxima*) and mussels (*Mytilus edulis)*. The remainder of the production is for rainbow trout (*Oncorhynchus mykiss)*.
Iceland	1300 (marine pens) 3800 (contained systems)	Atlantic cod (*Gadus morhua*), halibut (*Hippoglossus hippoglossus*) and turbot (*Psetta maxima)* in marine cages with Arctic char (*Salvelinus alpinus)* and Atlantic salmon (*Salmo salar*) in contained systems. There is increased interest in the farming of Atlantic halibut (*Hippoglossus hippoglossus*), sea bass (*Dicentrachus labrax*), turbot (*Psetta maxima*), and abalone (*Haliotis rufuscens*) and mussels (*Mytilus edulis)*.
Belgium	49 (2011)	Predominantly freshwater species such as rainbow and brown trout (*Salmo salar)*, common carp (Cyprinus carpio), tench (Tinca tinca), roach (Rutilus rutilus), pike (*Esox lucius) and* cichlids (*Sarotherodon niloticus, Tilapia aurea* and *T. hornorum*). Currently investigating commercial aquaculture of burbot (*Lota lota*) the only freshwater member of cod family.
Greenland	No data available from FAO	Aquaculture was attempted in Greenland in the 1980s, although aquaculture is not conducted at the present time.

The following sections covering the interactions between marine mammals and aquaculture and its management and mitigation has focused on those countries where greatest interaction has been noted, namely Norway, UK (Scotland) and Canada.

9.3 Interactions between aquaculture and marine mammals

In the North Atlantic, mammals often associated with aquaculture sites include two species of seals (*Halichoerus grypus* and *Phoca vitulina)*, otters (*Lutra lutra*), minks (*Neovison vison*), dolphins and porpoises (particularly *Tursiops truncatus* and *Phoceona phocoena*) (Northridge *et al.,* 2010). For the purposes of this review, it is seals and cetaceans that are focused upon. In the majority of instances, the associations between marine mammals and aquaculture sites are benign. Occasionally, however, the interactions can lead to conflict, with seals in particular having a direct impact on the industry.

The main interactions noted between marine mammals and aquaculture can be divided into: 1. gear damage and associated fish welfare issues, 2. entanglements in nets, ropes and moorings, and 3. disturbance and habitat exclusion. However, where the escapees, due to gear damage, are salmon, there can be serious biological consequences through the impact on the genetics of local wild fish stocks. This aspect is being reviewed by WGAQUA, 2014.

Managing the interaction between marine mammals and aquaculture will require a management strategy that upholds legal obligations to protect marine mammal populations, while minimising or eliminating damage to the aquaculture industry that is caused by marine mammals.

9.3.1 Damage to gear caused by marine mammals and associated fish welfare issues

In the close vicinity of aquaculture installations, most marine mammals, including seals, tend to concentrate their foraging on wild fish congregating outside farm cages that are taking advantage of fish feed that falls through cages (Dempster *et al.*, 2009; Diaz Lopez, 2012; Northridge *et al.*, 2013). For example, opportunistic feeding provides a reliable food source for the bottlenose dolphins year-round, although the occurrence of dolphins peaks in autumn when they naturally tend to stock up their fat stores, but also coinciding with a reduction in local prey availability (Diaz Lopez, 2012).

Seals do not necessarily focus their foraging activities in areas close to aquaculture installations but if a farm is located in close proximity to their haul-out site, they may take advantage of the available foraging opportunity (Nelson *et al.*, 2006; Northridge *et al.*, 2013). However, most individual seals at farm sites actually appear to be making use of locally abundant wild fish associated with the farm rather than targeting the farmed fish (Northridge *et al.*, 2013). Seals can, and do, attempt to prey on the fish through the cage netting (Wursig and Gailey, 2002; Quick *et al.*, 2004; Diaz Lopez, 2006), particularly salmon, killing or maiming fish by taking bites out of them or clawing at them. Seals can also breach the containing net, allowing fish to escape, sometimes in large numbers. Additionally, their presence around fish cages is said at times to frighten fish to an extent that they may stop feeding and fail to grow. This is a welfare issue for the fish, but is also an obvious economic issue for the farms themselves. Quantifying the scale of these impacts is difficult.

There are few records available in the public domain that quantifies the numbers of fish killed or injured by seals. However, in Scotland, the Scottish Aquaculture Code of Good Practice, revised in 2013 (see http://www.thecodeofgoodpractice.co.uk/publish), requires farms to keep records of dead fish ("morts"):

> 3.2.7 At all stages, the number of dead fish must be recorded, along with, where possible, a record of the cause of death.
>
> 5.2.9.4. Farmers should keep records of losses to predators and use of control systems.

Furthermore:

> 5.3.5.1 Fish should be inspected daily and dead or moribund fish should be removed, minimising handling to avoid stress to the live fish within the enclosure.
>
> 5.3.5.3 Records should be kept of each inspection, which include the number of dead fish removed and the likely cause of death, as determined by a competent person.

These data, if properly collated, can provide a great deal of information on seal interactions with salmon at farm sites. Preliminary analysis by Northridge *et al*. (2013) identified 4 types of mort damage based on photographic evidence (Figure 9.1). This allowed general inferences about the aetiology of the damage, for example, that wounds of categories 2 and 4 fish are inherently different and have almost certainly been caused by very different forms of attack. It seems likely that obtaining an understanding of the difference between these attack strategies is key to designing effective anti-predator measures. The techniques developed by Northridge *et al*. (2013) require further refinement before they can be used to make useful inferences about the number of animals involved in an attack, or the particular species involved. Such development will require contributions from industry but, unfortunately, these have not been forthcoming to date. Without assistance and encouragement from management within the industry, fish-farm workers have no practical and immediate reason to contribute to such investigations.

Type 1: Damage: Spine and head left

Type 2: Tail removed through meshes

Type 3: Multiple parallel gashes - possible flipper damage

Type 4: Typical "belly bites" from larger salmon

Figure 9.1. Mort damage associated with seal predation (taken from Northridge *et al*., 2013).

A separate issue is the damage caused to the gear from animals trying to chew through the nets, which then leads to fish escaping (Figure 9.2). The impact of this varies depending on place and gear types. Overall, the loss of fish due to marine mammal incidents is considered to be relatively low (approximately 2–5%) in most countries, although in some areas, notably Scotland, it has been reported as being as high as 27% (NASCO, 2007; Thorstad *et al.*, 2008). In Scotland, the escapes of fish that are attributable to seals can be quantified because under the Registration of Fish Farming and Shellfish Farming Businesses Amendment (Scotland) Order 2008 (and also under Section 4.10 of the Scottish Finfish Farming Code of Good Practice) any fish escaping from salmon farms must be notified to the Scottish Government within 24 hours of discovery. Companies are also obliged under the Fish Farming Businesses (Record Keeping) (Scotland) Order 2008 to maintain specific records relating to fish containment and breaches of containment, including details of net and mooring types as well as any anti-predator measures undertaken. In 2011 ten Atlantic salmon escape incidents were notified, involving 403 000 fish. Three of these incidents involving 21 000 fish were recorded as having been caused by predators; presumably seals (Northridge *et al.*, 2013). Note, however, that this aspect of seal damage is likely to be the least significant commercially, considering some farm sites are reported to have almost daily losses of fish to seals as described above (Taylor and Kelly, 2010a,b; Northridge *et al.*, 2010).

Figure 9.2. Typical predator hole in a net (photo credit: Knox Nets; taken from Northridge *et al.*, 2013).

Quantifying the extent to which seals may scare fish, and so affect their growth, is difficult and, to date, there appear to be no studies that have quantified this. In Scotland, under Part 6 section 110 of the Marine (Scotland) Act 2010, fish farms that profess a need to shoot seals must apply for a licence to do so, in order to "protect the health and welfare of farmed fish". Analysis of licence applications and returns may lead eventually to a better understanding of the extent to which fish welfare is affected by seals giving some impression of the scale of welfare concerns (see Section 9.4.4 for further details).

9.3.2 Marine mammal entanglements in nets, ropes and moorings

When air breathing mammals operating in the close vicinity of nets, ropes and moorings around a farm, there is a risk of entanglement. However, there have only been a few documented cases (EW, 2008). Regarding cetaceans, Kemper *et al*. (2005) reported bottlenose dolphins (*Tursiops aduncus*) and common dolphin (*Delphius delphis*) becoming entangled in bluefin tuna pens in South Australia. Diaz Lopez and Shirai (2007) recorded the entanglement of three bottlenose dolphins (*Tursiops truncatus*) in Italy over a 15 month period. Entanglements have also effected larger species, such as grey whales (*Eschrichtius robustus*) which have become entangled in herring pens (Wursig and Gailey, 2002), a humpbacked whale (*Megaptera novaeangliae*) entanglement occurred in salmon farm in Norway (Arne Bjørge, pers. comm.) and Canada (DFO, 2014) and two Bryde's whales (*Balaenoptera edeni*) have become entangled in mussel spat collection ropes in separate incidents in New Zealand (Wursig and Gailey, 2002; Loyd, 2003).

Regarding seals, the number of reported deaths caused by entanglement is unsurprisingly higher than for cetaceans. In Canada, 13 harbour seals, six California sea lions and one unknown species were recorded in 2012 (DFO, 2014). In the UK, although the cause of death is determined where possible in marine mammals through the UK CSIP, e.g. in ten grey seals and three harbour seals drowning/entanglement are listed as cause of death, currently there is no way to separate drowning/entanglement in aquaculture gear from that associated with fishing gear (Andrew Brownlow, pers. comm.).

9.3.3 Disturbance and habitat exclusion

The use of sound through acoustic deterrent devices (ADDs) often referred to as 'seal scarers' has been utilised in many Countries to reduce seal predation on farmed fish (Würsig and Gailey, 2002; Janik and Gotz, 2013). There are several ADD manufacturers who sell or lease devices that are intended to shock, scare or be unpleasant or painful enough to the target animal, usually seals, and to keep them at distance from the pens (Würsig and Gailey, 2002). ADDs have been used to reduce predation at marine salmon farms at least since the 1980s (Graham *et al*., 2009). Quick *et al*. (2002) found that 52% of Scottish managers participating in their questionnaire utilised seal scarers, making this method of control the most common against underwater predation.

The acoustic signals from such devices can be detected at more than 14 km from the sound source, although propagation losses are site-specific and variable (Northridge *et al*., 2010). The increasing use of acoustic deterrent devices (ADDs) to deter predators from aquaculture installations has caused concerns about the effect on both targeted and non-targeted animal species. Marine mammals are particularly sensitive to high intensity sounds and, consequently, this method could be harmful to the animals by causing a temporary threshold shift or even permanent hearing damage if animals are exposed for long enough or are habitually within very close distances to the transducers (Gordon and Northridge, 2002; Schakner and Blumstein, 2013). The mid- to high frequencies used are also within the sensitive hearing range of odontocetes, and several studies have demonstrated that cetaceans, especially harbour porpoises are to some extent excluded from areas around farm sites where they are being used (Olesiuk *et al*., 2002; Johnston, 2002; Northridge *et al*., 2010). However, Northridge *et al*. (2010) noted that whilst avoiding areas with active ADDs, harbour porpoises were recorded feeding approximately 200 m from active ADDs suggesting that the exclusion effect is voluntary rather than obligatory. Harbour porpoises also immediately returned to areas when the ADDs were switched off. Northridge *et al.* (2010) also reported that harbour porpoises appeared to avoid an area where ADDs had recently been installed but were less averse to another area where ADDs had been used for several years. Habitat exclusion effects have also been noted in killer whales (*Orcinus orca*) near salmon farms utilising ADDs, with sightings increasing six months after use of ADDs stopped (Morton and Symonds, 2002). In contrast, however, Diaz Lopez and Marino (2011) found that bottlenose dolphins were not excluded from an area with use of ADDs. Acoustic signals vary depending on the topography of the area and the use of these devices and their effect on surrounding wildlife may be individual from site to site (Gordon and Northridge, 2002; Northridge *et al.*, 2010).

Issues have also been noted with respect to shellfish culture. In the Netherlands, the effect of location of mussel seed collection devices on the local seal colonies in the Wadden Sea has been modelled (Cremer *et al.*, 2012). There was a negative correlation between the number seals present and the number of mussel seed collecting devices whilst the seal colony growth rate of in three areas with these devices was lower than the growth rate in areas without mussel seed collection devices. Additionally, these devices are constructed by piling and, thus, potentially having effects on marine mammals during construction. De Haan and Burggraaf (2012) noted that there was a no influence during piling beyond 25 m of sound source (through air). Temporary threshold shift (TTS) was reached at 100 m from the sound source, with behavioural changes in seals being possible at more than 10 km. For porpoises, TTS was reached at 850 m from single sound source, with behavioural changes noted to 10 km. When two piles were working at the same time, TTS increased to 1600 m.

9.4 Management to reduce conflict between aquaculture and marine mammals

As marine mammals, particularly seals, can have a negative impact on aquaculture, extensive effort has been put in place, both by farmers and by researchers, to reduce the conflict. The most important measures are good husbandry practices which include maintaining nets in good condition, removing dead fish as soon as possible and ensuring nets are adequately tensioned, as well as lower stocking densities and larger cages.

A range of additional measures have been or are being used, including anti-predator nets (Diaz Lopez, 2012), ADDs (Würsig and Gailey, 2002; Janik and Gotz, 2013) and lethal control through legal or illegal shooting (Thompson *et al.*, 2007; EW, 2008). In the past, trapping and relocating problem individuals (Pemberton and Shaughnessy, 1993), methods such as feeding distasteful or emetic foods (Würsig and Gailey, 2002) and playing vocalisation sounds from killer whales (*Orcinus orca*) (Deecke *et al.*, 2002) have all been trialled but have not proven successful at the commercial scale.

9.4.1 Good husbandry practices

The development of protocols and best practice guidelines for containment have been published for Scotland, although the possibility of making anything other than general recommendations for controlling seal predation was not possible (Thistle Environmental Partnership, 2012). This report deals with many of the technical aspects of cage design, and makes several recommendations for further research, some of which are focused on *"Whilst farmers had opinions and evidence about the nature of seal attacks, this was based on observation only and was not quantifiable or objective. It appeared that farmers did not have access to any research in regard to seal predation."*

Tensioning nets in aquaculture cages is essential to maintain net volume and structure, especially at high tidal energy sites. Net tensioning is also widely cited as being a critical issue in minimising seal depredation. The rationale here is that nets that are poorly tensioned may provide the opportunity for seals to makes holes more easily, may enable seals to deflect netting to a greater extent to grab fish within a cage, and/or may result in loose folds of netting that trap or restrict the movements of fish, enabling seals to grab them. In fact none of these rationales has been demonstrated, but there is nevertheless a widespread assertion that a taught net helps minimise seal depredation.

Taylor and Kelly (2010a) recommended that research should be conducted into the role of the strength and construction of cages in deterring predator attacks. In this context, Taylor and Kelly (2010a) reported that some manufacturers claim that high modulus polyethylene (HMPE-including Dyneema™) nets are "more resistant to predators". Others suggest that PVC coated nets (Aquagrid™) will also "virtually eliminate fish loss due to attacks and escapes". There are several other net types (e.g. Sapphire netting from Garware, India) that have also been described as being predator resistant. Some have steel or copper cores. Many such nets have been deployed and tested in Scotland but without any over-arching coordination or impartial review of how they have worked. Much of the discussion on new netting materials appears to focus on their increased strength, which makes breaking the meshes much harder for a predator, and which will therefore minimise fish loss through escapes due to holes in the net. However, most seal damage is caused by seals biting fish through the meshes without causing holes. The extent to which any new type of netting may inhibit such behaviour is not clear.

Indeed, in developing draft protocols for containment, TEP (2012) identified several of these issues relating to predation as being major knowledge gaps. In relation to the development of a Scottish Technical Standard (STS) on containment, they concluded as follows:

> It is not proposed to include specific net measures from a design and construction perspective to protect against predation, since there is considered to be insufficient objective evidence on which to base such measures. Additional research is required on the way in which different freshwater and seawater predators breach net integrity and how effective possible defence measures might be. Whilst not essential to publishing a STS, it is highly desirable to include predation at the earliest opportunity as it is such an important issue in Scottish finfish farming. Although there is knowledge of seal 'attacks' at seawater sites most, if not all, appears to be anecdotal. Whilst all farmers consider that net tensioning is effective, there appears to be no information on how tight such tensioning should be and whether higher net strengths (or indeed net materials) may provide greater resistance.

9.4.2 Anti-predator nets

Anti-predator nets are widely used in most fish producing nations, although not Scotland. Such nets are designed to keep predators away from the farming pens to reduce fish mortality by marine mammals. These may be curtain type nets that surround each pen or surround the entire site, and which hang to the bottom of the seabed, or box-type, where netting extends under the site too (Northridge *et al.*, 2013). Other tactics employed by fish farmers include false bottom cages where additional netting under each pen ensure that fish are less accessible to attack from below, and seal blinds where that part of the net where dead fish accumulate has fine meshed net stitched into it to try to make any dead fish less visible and so discourage seals from learning how to attack fish through the netting.

Such approaches can be effective if properly maintained, although not 100%, and they are expensive. Concerns have been raised regarding a possible reduction in water flow to the cages with the use of such nets, which impacts on water quality (Northridge *et al.*, 2010). There are also accounts of mammals and birds becoming entangled in such nets, and seals may learn how to breach them (Northridge *et al.*, 2013). The large-meshed predator nets with a mesh size of 10–15 cm were considered to cause the most concern as such nets tend to be looser in structure and have an increased risk of entanglement over smaller mesh sizes (Diaz Lopez, 2007; Northridge *et al.*, 2013). Exterior nets also require additional mooring and these and the nets themselves can pose problems for boats tending and working around the farm site (Northridge *et al.*, 2010; Northridge *et al.*, 2013). Anti-predator nets need to be designed and set properly if entanglement issues are to be reduced.

9.4.3 Acoustic methods

There is still uncertainty surrounding the extent to which of ADDs are effective in reducing pinniped depredation at farm sites (e.g. Nelson *et al.*, 2006; Northridge *et al.*, 2010). Few data are available, and this is an area that urgently requires further research if the obligations of the Habitats Directive are to be properly addressed by EU Member States due to concerns that such devices have a disproportionate effect on cetaceans. There is evidence that ADDs can be effective at least in the short to medium term at salmon trapnet fisheries (Fjälling *et al.*, 2006; Graham *et al.*, 2009; Harris *et al.*, 2014), but the context and behavioural state of seals may differ from those at farm sites. No study has demonstrated the effectiveness or otherwise of ADDs used at salmon/fish farms (Northridge *et al.*, 2013). Northridge *et al.* (2010) reported on the basis of site interview that: "there was much equivocation about the effectiveness of ADDs. Most people that used them reckoned that they reduced seal attacks without eliminating them, and at 15/20 sites they were judged overall to have some preventative effect, and not at five". Likewise Taylor and Kelly (2010a) reported that "many consultees commented that ADD appeared effective to start with and then seal became used to them".

ADDs can have a significant impact on cetacean distribution with reduced porpoise detections within several kilometres of active ADDs (e.g. Olesiuk *et al.*, 2002; Johnston, 2002; Northridge *et al.*, 2010). However, all of these studies have used the same type of ADD, Airmar, made by a single manufacturer). Northridge *et al.* (2013) conducted a series of trails with a different ADD with strikingly different results. Harbour porpoises showed weak or minimal responses to the sounds generated by the Terecos device. Northridge *et al.* (2013) recommended that further tests using Terecos ADDs and those of other manufacturers would help to ascertain the extent to which these results can be generalised.

Under the Global Standards for Salmon Aquaculture, initiated by the WWF and agreed by over 500 international stakeholders, ADDs are intended to be phased out in salmon aquaculture within three years of the publication of the Salmon Aquaculture Dialogue, which occurred in 2012 (SAD, see http://assets.worldwildlife.org/publications/433/files/original/SAD_Standard_Final_Draft.pdf?1346188051). The SAD proscription of ADDs is based on the assumption that all such deterrents are inimical to cetacean conservation. An exception to this may be granted where new technologies can be shown to present less risk to non-target populations. For example, Janik and Gotz (2013) tested a startle reflex ADD prototype for

13 months at salmon farms in Scotland. The study found that the loss of fish was significantly reduced after the instalment of the ADD compared to the pre-deployment period. There were sightings of seals, porpoises and otters close to the farm during the study period, however, the presence of these animals did not result in observations of any additional predation, and it was therefore concluded that the ADD was effective against seal predation without excluding harbour porpoises from these areas (Janik and Gotz, 2013).

9.4.4 Deliberate killing of marine mammals

A common method of controlling the predation issue is non-lethal or lethal shooting (Würsig and Gailey, 2008; EW, 2008; Graham *et al*., 2011a). Warning shots can be used to encourage the animals to leave the area, however, it is not uncommon for animals to be shot dead when they venture into the vicinity of the farms (Pemberton and Shaughnessy, 1993; Quick *et al*., 2002; Butler *et al*., 2008; Graham *et al*., 2011a).

Grey and harbour seals are listed on the Norwegian red list for threatened species, with harp and ringed seals also occurring in Norwegian waters. In 2010 the status of grey and harbour seals was considered to Least Concern and Vulnerable, respectively (Artsdatabanken, n.d-a; Artsdatabanken, n.d-b). The Forskrift om regulering av sel på norskekysten Regulation (translating as regulating seals along the Norwegian coast based on the Marine Resource Act) states that seals that damage fishing gear or aquaculture instalments can be killed by owner, user or other person with connection to the equipment or instalment. Euthanasia should only occur where reasonable attempts to use other mitigation options have been implemented (FiskeriDir, n.d). Additional hunting is authorised outside the pupping season for grey, harbour, harp and ringed seal and controlled through an annual quota system. There are restrictions on the types of firearms to be used and hunters are required to hold a current licence, which includes an annual shooting test. All catches are to be reported to the authorities.

In Scotland, as the conservation concern rose over declining population numbers of harbour seal, there has been increased concern about the shooting of seals from aquaculture farms. Harbour seals are listed on the Annex II of the EC Habitat Directive and are also an important species for tourism interests. These conflicting concerns lead to a revision and introduction of new legislation. Previously, the UK's Conservation of Seals Act 1970 (CoSA) was used to manage the interaction between seals and fish farms and netting stations. This act allowed for shooting of seals year-round, except during the pupping season. There was no requirement to obtain a licence to shoot seals or to report the number shot. In 2010, the Marine (Scotland) Act 2010 came into force which seeks to balance seal conservation with sustainable fisheries and aquaculture. The Act made it '*an offence to kill or injure a seal except under licence or for welfare reasons, outlawing unregulated seal shooting that was permitted under previous legislation*'. Additionally, seal conservation areas were introduced which are designed to protect the vulnerable and declining harbour seal.

The introduction of the licensing system has led to a significant reduction in the shooting of seals year on year. Licences are granted for aquaculture to shoot seals based on a licence application which details levels of damage in recent years, and total allowed removals are limited by Potential Biological Removal (PBR) calculations for each of seven seal management areas in Scotland. Where a licence is granted, the marksmen must be nominated and have proven adequate skills and experience in using firearms.

The results in terms of licence applications, numbers granted and the actual numbers of seals shot are published annually. The latest figures available are for 2013 in which permission was granted to shoot 774 grey seals (0.7% of population) and 265 harbour seals (approximately 1% of minimum population estimate). Of these 238 grey seals and 36 harbour seals were reported as having been shot (see Tables 9.2 and 9.3). The maximum number of seals allowed on licences in 2014 is 765 grey seals and 240 harbour seals. This represents less than 0.7% of the grey seal population and approximately 1% of the minimum harbour seal population (Scottish Government, 2014b).

Table 9.2. Comparison of the number of licences applied for, number granted and number of grey seals shot by seal management area.

Seal Management Area	Applied for	PBR	Seals granted	Seals shot
East coast	142	314	82	28
Moray firth	145	174	90	43
Orkney & North Coast	355	1448	220	87
Shetland	240	236	105	54
Southwest Scotland	63	57	26	1
Western Isles	198	387	125	10
West Scotland	204	386	126	15

Table 9.3. Comparison of the number of licences applied for, number granted and number of harbour seals shot by seal management area.

Seal Management Area	Applied for	PBR	Seals granted	Seals shot
East coast	54	2	0	0
Moray firth	34	17	16	3
Orkney & north Coast	37	17	5	1
Shetland	23	18	6	3
Southwest Scotland	88	35	30	0
Western Isles	75	82	45	1
West Scotland	291	446	163	28

In Canada, the Department of Fisheries and Oceans Canada (DFO) are responsible for the management of marine mammals. Prior to 2010, Nuisance Seal Licences for aquaculture facilities were issued under the Marine Mammal Regulations. Conflicts between marine mammals and aquaculture are more prevalent on Pacific coasts where the greatest concentration of the industry is sited. The number of animals killed annually is recorded and had been a declining trend in more recent years. The numbers decreased from 577 harbour seals in 1995 to 56 in 2010, and 243 California sea lions in 2000 to 170 in 2010. The *Pacific Aquaculture Regulations* were implemented in 2010, which led to a new licence system for aquaculture sites. All facility operators have to have a Predator Management Plan in place and non-lethal measures to deter and minimise interaction between the farms and the marine mammals should be implemented. The most common system to deter marine mammals is now the use of anti-predator nets that usually surround the entire farm structure. As a last resort, if harbour seals or Californian sea lions represent imminent danger to the aquaculture facility or staff, they may be lethally removed under licence. In 2011 and 2012, 36 and five harbour seals were lethally removed, respectively whilst for California sea lions the numbers were 141 and four (DFO, 2014). For other marine mammal species, an additional licence has to be obtained due to the conservation status of these species under the Species at Risk Act (SARA) (DFO, 2014).

9.5 Summary

Managing the interaction between marine mammals and aquaculture will require a management strategy that upholds legal obligations to protect marine mammal populations while minimising or eliminating damage to the aquaculture industry that is caused by marine mammals. As marine mammals, particularly seals, can have a negative impact on aquaculture, extensive effort has already been put in place, both by farmers and by researchers, to reduce the conflict. The most important measures are good husbandry practices which include maintaining nets in good condition, removing dead fish as soon as possible and ensuring nets are adequately tensioned, as well as lower stocking densities and larger cages. The most common other measures employed are the use of acoustic deterrent devices (ADDs) and lethal control, although the use of both of these is either being phased out or is tightly controlled in most countries where there is a significant conflict.

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10 ToR H Special Request: Marine Mammals (OSPAR 6/2014)

Advise on appropriate management units (MUs) for grey and harbour seals in the OSPAR Maritime area;

Provide technical and scientific advice on options for ways of setting targets for the OSPAR common MSFD Indicators for marine mammals and where possible, provide examples of the application of these options. The advice should consider the suitability of various options for relevant marine mammal species/ MUs/ indicators. In considering target setting options, also consider the consequences that this may have for the monitoring programme (including spatial and temporal implications). Consideration should be given to precision in target setting and monitoring. (Note that ICES are not asked to make any societal/ policy choices, but if necessary should identify the need for such choices and their potential implications);

Provide an overview of existing monitoring per OSPAR common MSFD indicator and marine mammal species, including the description of current monitoring frequency (and whether this is likely to be sufficient to meet the assessment requirement);

Provide an overview of possible future monitoring requirements and methodology per OSPAR common MSFD indicator and marine mammal species.

The request is to cover OSPAR regions II, III and IV.

The existing indicator technical specifications developed by COBAM should form the basis of this work.

Proposed OSPAR COBAM common marine mammal indicators are (see Annex 3 for full description):

CODE	PREVIOUS CODE*	INDICATOR	CATEGORY
M-1	31&33	Distributional range and pattern of grey and harbour seal breeding and haul-out sites, respectively	Core
M-2	32&34	Distributional range and pattern of cetaceans species regularly present	Core
M-3	35	Abundance of grey and harbour seal at breeding and haul-out sites, respectively	Core
M-4	36	Abundance at the relevant temporal scale of cetacean species regularly present	Core
M-5	37	Grey seal pup production	Core
M-6	38&39	Numbers of individuals within species being bycaught in relation to population	Core

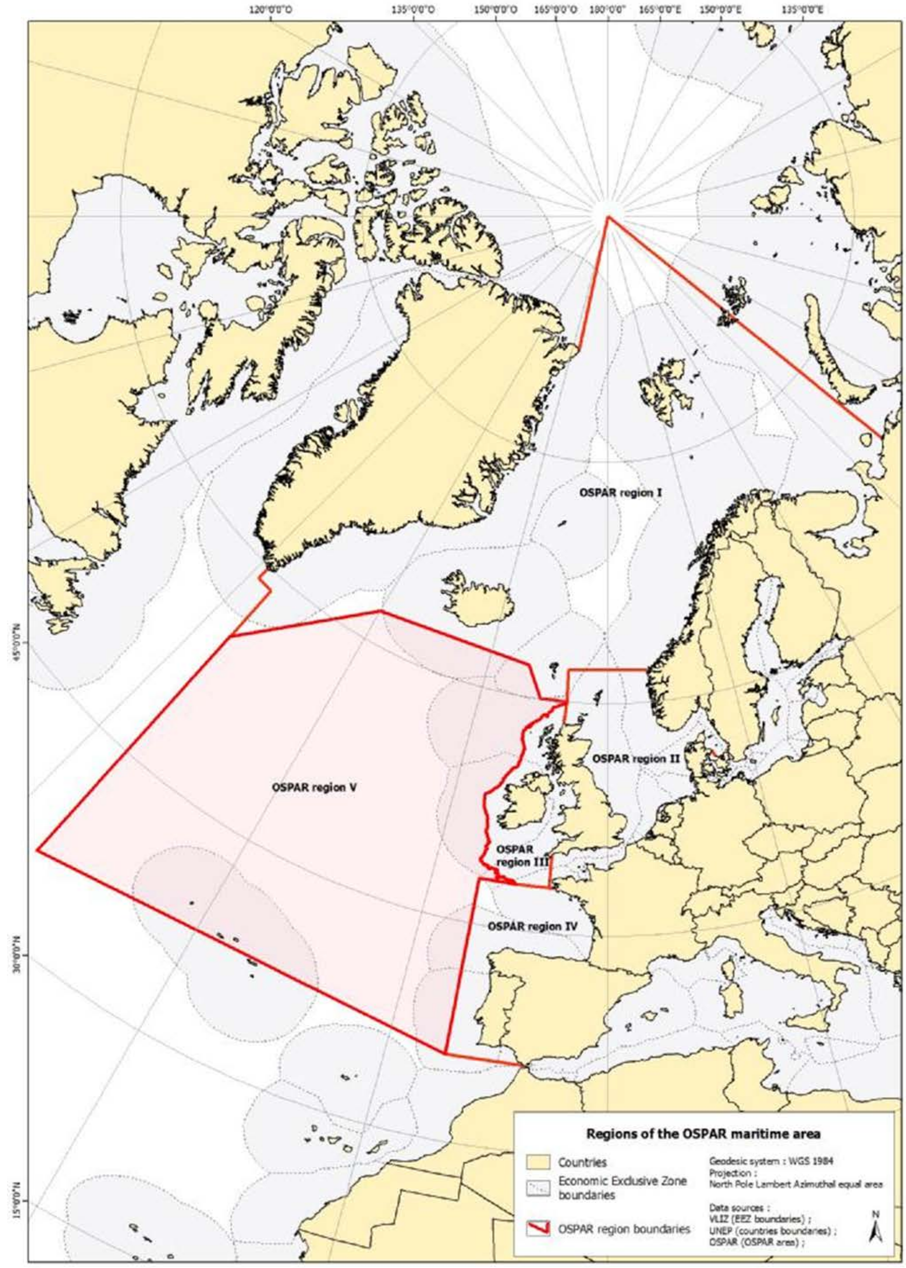


Figure 10.1. Regions of the OSPAR maritime area and Economic Exclusive Zone boundaries.

As indicator M-6, number of individuals within species being bycaught, is addressed in ToR c (Section 5), this indicator will not be addressed within this ToR. Due to a lack of agreed conservation objectives, decisions on mechanisms for determining safe bycatch limits (and production of those limits) is currently being stalled.

In 2009, ICES advised the European Commission 'that a Catch Limit Algorithm approach is the most appropriate method to set limits on the bycatch of harbour porpoises or common dolphins. In order to use this (or any other) approach, specific conservation objectives must first be specified. In both species improved information on bycatch and the biology of the species would improve the procedure.'

This was reiterated in 2010 'ICES advised in 2009 of the need for explicit conservation and management objectives for managing interactions between fisheries and marine mammal populations. This advice has not been acted upon. Lacking these objectives, ICES is unable to properly consider the impacts of these interactions in its management advice.'

In 2013, in the absence of action and in the context of the development of MSFD targets for marine mammal bycatch, WGMME had again strongly recommended that this advice was acted upon.

As part of the reforms to the Common Fisheries Policy and the Data Collection Framework, the European Commission requested that ICES provide advice on the use of management frameworks and other mechanisms for determining safe bycatch limits in 2013. This request was dealt with by WGBYC rather than WGMME. The ICES response noted that further work in this area would be required and that: 'This could be in the form of a workshop for invited participants representing managers, scientists and stakeholders. As stressed in the advice, input from management and from the "societal" side is crucial to such a process. We would envisage attendees from relevant parts of the European Commission (at least DG Mare and DG Environment), Member State fisheries authorities, the RACs, relevant intergovernmental bodies (Regional Seas Commissions, ASCOBANS and ACCOBAMS) and relevant NGOs. Whether the workshop should be arranged in Brussels, in ICES or elsewhere depends on an assessment of how to ensure the best input from the policy/management and societal sides.

Before the meeting it is essential to ask scientists to prepare presentations of models and some model scenarios which can be used during the workshop. Dependent on the timing of the workshop such input could be prepared by ICES expert groups (i.e. WGBYC and WGMME) leading to a workshop in late summer 2014 or alternatively by a smaller invited expert group if the Commission wants the workshop in autumn 2013. We are aware of some relevant preparatory work being done under funding from UK and from ASCOBANS that will be ready in August 2013 for consideration by an ASCOBANS meeting and aims to complete by the end of September 2013.'

The European Commission have yet to respond to ICES regarding this offer. Consequently, WGMME **recommends** that European Commission give serious consideration to ICESs offer to host a workshop, with the objective of reviewing different mechanisms for determining safe bycatch limits and finalising conservation objectives for a bycatch limit approach that would enable conservation aspiration to be met.

Until the European Commission have determined the approach they will use for deciding safe bycatch limits, and have also finalised reform of the CFP, the DFC and possibly Regulation 812/2004, it is not sensible to develop guidelines and protocols to aid MS in meeting their MSFD obligations.

10.1 Assessment units for seals and cetaceans

For the purposes of this OSPAR request, WGMME defined a management unit (MU) as typically referring to animals of a particular species in a geographical area to which management of human activities is applied. A MU may be smaller than what is believed to be a population to reflect differences in spatial preferences of individuals (an ecological unit, see (Evans and Teilmann, 2009; Evans, 2012a)) and/or spatial differences in human activities. However, what matters in the context of management is whether human activities could impact individuals from different populations differentially if no structure were imposed by management units. For example, if fisheries bycatch of a particular species were concentrated in an area to which individual animals had a preference to return over a period of their lifetime, this may lead to local depletion in that area which could be justified as a MU. However, if individuals replaced the removed animals quickly, there may be no local impact and no separate MU would be necessary. If MUs are defined to be smaller than a population, it is important that management takes into account the rates of interchange of individuals between MUs; that is, the MUs should not be treated as if they were demographically independent.

At times, there are some ambiguities between research scientists and managers on the actual definition and use of the term 'management unit' or MU. For clarification purposes, the designation 'assessment unit' is proposed instead of 'management unit' for marine mammal species included in MSFD indicator assessments. Within HELCOM, the designation 'assessment unit' is also used for marine mammals (Härkönen *et al.*, 2013).

10.1.1 Proposed Harbour Seal Assessment Units for OSPAR Regions II, III, IV

In 2009, the WGMME reviewed the geographical EcoQO subunits for harbour seals (*Phoca vitulina*) in the North Sea, taking into account biologically appropriate management units (MUs). Since then, two genetic studies were undertaken including Islas-Villanueva *et al.* (2012), which was reviewed by the WGMME in 2012, and Olsen *et al.* (2014). Olsen *et al.* (2014) proposed a northern Skagerrak and a southern Skagerrak management unit, thus splitting the Skagerrak and Oslo fjord EcoQO subunit into two. In contrast results from Islas-Villanueva *et al.* (2012) supported the management units defined for harbour seals in Scotland. Although some broader genetic clustering was apparent, the structuring based on haul-out sites and associated local foraging areas is likely to be as important in the management of these populations as the maintenance of their genetic diversity (ICES WGMME, 2012).

Unlike grey seals, harbour seals tend to undertake relatively short excursions from their favoured haul-out sites, often less than 50 km (although they may range over much larger distances) and there is little evidence of extensive seasonal migrations (ICES WGMME, 2009).

Figure10.2. Proposed seal management units around the UK.

It is recommended that EcoQO subunits for harbour seals in the North Sea should be retained (taking account of previous proposals for alterations to these, see WGMME 2009, 2012 and 2013) and employed as assessment unit (AUs) for this species within the MSFD indicator assessments. Around the UK, MUs have been delineated for harbour seals based on the locations of breeding colonies and haul-out sites, and on administrative boundaries (see Figure 10.2). It should be noted that in all but one case (Northeast England MU), the new UK harbour seal management units in the North Sea are comprised of previously defined EcoQO subunits (see Table 10.1). For ease of reporting, these new UK Management Units will be used within MSFD indicator assessments. Analysis of telemetry data of harbour seal movements suggested that projection of UK seal MU boundaries covering the extent of UK territorial waters would be prudent, and in alignment with the approach taken for other marine mammals in the UK (Hanson and Lonergan, 2012). This work is ongoing at present, with completion expected later in 2014. Harbour seals in French waters of the North Sea and Channel should be assessed as one separate assessment unit (ICES WGMME, 2013). Telemetry work undertaken to date in the three main colonies suggests that harbour seals are very coastal, staying within 100 km of their haul-out site (Cecile Vincent, pers. comm.).

Ireland has not yet proposed specific management units/areas but the Department of Arts, Heritage and the Gaeltacht has reviewed OSPAR/ICG-COBAM proposals, and is considering a UK-type 'subunit' approach, i.e. regional segmentation with each segment covering both seal species, although it is mindful of (i) the current absence of a genetic basis for the delineation of distinct units/subareas but also of (ii) the movement patterns shown by each species based on national/international research results. Possibly regional units could be: (1) East/Southeast, (2) Southwest, (3) West, and (4) Northwest. This division would capture the main national regional population centres, which are comparatively isolated from each other. Figure 10.3 provides a summary of the proposed AUs for harbour seal.

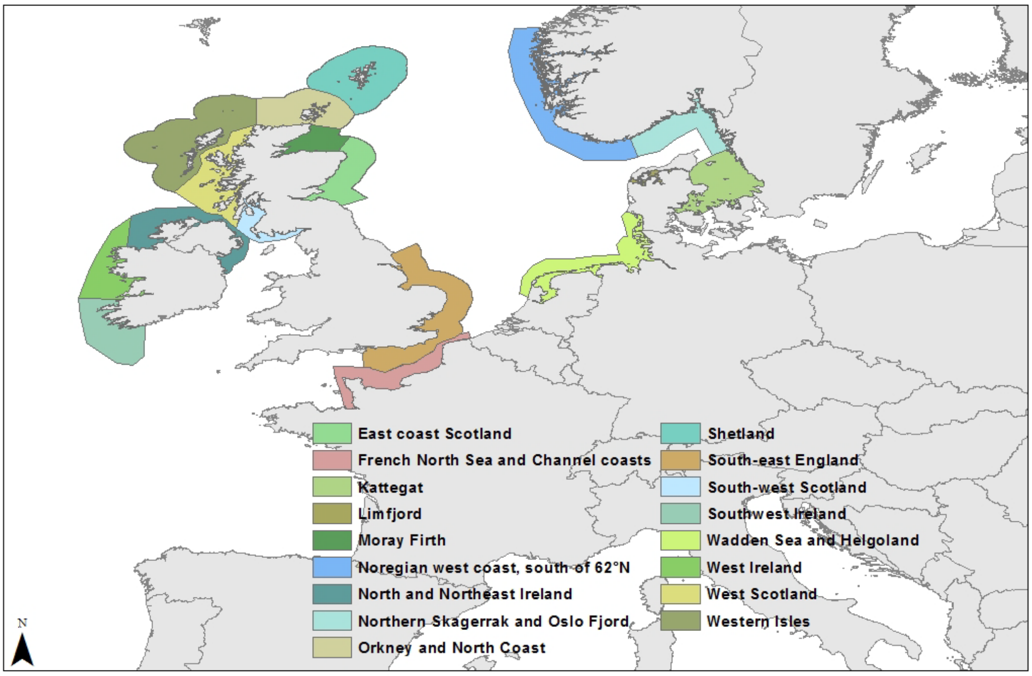


Figure 10.3. Proposed harbour seal assessment units.

Table 10.1. Proposed assessment units for harbour seals in the OSPAR region II, III and IV.

Country	Assessment unit	Reference
The Netherlands	Delta area (although numbers very low)	EcoQOs subunit (ICES WGMME 2009)
Netherlands, Denmark & Germany	the Wadden Sea	EcoQOs subunit (ICES WGMME 2009)
Germany	Helgoland	EcoQOs subunit (ICES WGMME 2009)
Denmark	eastern Limfjord	EcoQOs subunit (ICES WGMME 2009)
Norway, Sweden	the Northern Skagerrak and Oslofjord	EcoQO subunit (ICES WGMME 2013); Olsen *et al*. (2014)
Sweden	Southern Skagerrak	Olsen *et al*. (2014)
Denmark and Sweden	Kattegat	EcoQOs subunit (ICES WGMME 2009); Olsen *et al*. (2014)
Norway	the west coast south of 62°N	EcoQOs subunit (ICES WGMME 2009)
UK	Shetland, Scotland	UK proposed Management Unit /EcoQO subunit
UK	Orkney and North Coast, Scotland	UK proposed Management Unit /EcoQO subunit
UK	Moray Firth, Scotland	UK proposed Management Unit /EcoQO subunit northeast Scotland
UK	East Coast Scotland	UK proposed Management Unit /encompasses EcoQO subunit southeast Scotland (Firth of Tay)
UK	Northeast England	UK proposed Management Unit /no previous EcoQO subunit
UK	Southeast England	UK proposed Management Unit /encompasses EcoQO subunit the Greater Wash
UK	West England and Wales	UK proposed Management Unit
UK	Southwest Scotland	UK proposed Management Unit
UK	West (Highland) Scotland	UK proposed Management Unit
UK	Western Isles, Scotland	UK proposed Management Unit
UK	North Ireland	UK proposed Management Unit
France	North Sea and Channel coasts	(ICES WGMME 2013)
Ireland	(to be confirmed)	

10.1.2 Proposed grey seal assessment units for OSPAR Regions II, III, IV

For grey seals, a considerable amount of movement occurs (observed using telemetry data) among UK MUs/EcoQO subunits in the North Sea (Hanson and Lonergan, 2013). Previous genetic studies reported differentiation between North Rona, NW Scotland and the Isle of May in eastern Scotland using microsatellite analysis (Allen *et al*., 1995), and clear genetic distinction between grey seals breeding in the southwest UK (Devon, Cornwall and Wales) and those breeding around Scotland and in the North Sea (SMRU, unpublished data).

Grey seals range widely at sea and may visit multiple distant haul-out sites (McConnell *et al*., 1999). Studies using flipper tags have indicated that young seals disperse widely in the first few months of life. Pups marked in the UK were recaptured or recovered along the North Sea coasts of Norway, France and the Netherlands, mostly during their first year of life (Wiig, 1986). Though individual mature seals of both sexes are usually faithful to particular breeding sites, and may return within 10–100 m of previous breeding sites (Pomeroy *et al.*, 1994; Pomeroy *et al.*, 2000; Hammond *et al.*, 2008).

The UK grey seal population represents approximately 38% of the world population on the basis of pup production (SCOS, 2012). There is no evidence to suggest that grey seals on the North Sea coasts of Denmark, Germany, the Netherlands, or France are independent from those in the British North Sea. Within this region, breeding sites are primarily located in Germany (Helgoland, Wadden Sea) and the Netherlands. However, abundance growth rates in excess of 20% in the Wadden Sea show that there is a substantial amount of immigration to this area (TSEG, 2012). Currently, grey seals numbers are also rapidly increasing, +25%/year, along the French Channel coast and, based on telemetry data of seals tagged in France, movements of animals from the eastern English Channel into the North Sea have been observed (ICES WGMME, 2013) (see Figure 3).

It is therefore recommended that AUs at a larger spatial scale within the North Sea would be more appropriate to MSFD indicator assessments. To assist such an approach, monitoring of grey seal pup production in the UK, Netherlands and Germany should follow guidelines outlined within the quality assurance statement (see Section 10.5). Moult counts, which serve as the primary monitoring in the Wadden Sea, are not comparable to results from UK estimation of pup production in grey seals; which constitutes the large majority of the abundance of grey seals in the North Sea AU. Thus, exact counts or estimates of pup production with coefficients of variation from the eastern North Sea are essential to monitoring/MSFD indicator assessments. The southern extent for the grey seal North Sea assessment unit includes the boundary with ICES Division V11e, and the northern extent should include the western Danish coastline in the east and the North Sea coast of Norway in the northeast. The northwestern extent should align with the UK's Orkney and north coast MU.

Telemetry studies of grey seals tagged at haul-out sites along the northwest coast of France have shown clearly that animals found in French waters are part of the same population found off western Britain and Ireland (ICES WGMME, 2013, see Figure 10.4). Grey seals tagged in Brittany, France hauled-out along the west coast of Ireland (especially in the Blaskets, the Inishkeas, etc.), and this was also confirmed by visual observations (Cecile Vincent, pers. comm.). Grey seals tagged along the western and southern Irish coasts moved into waters off western Scotland, Wales and southwest England (see Figure 10.5). Grey seals tagged on the west coast of England and Wales moved widely around the Irish and Celtic Seas, hauling out on Irish coasts, and animals tagged on the west coast of Scotland ranged into waters off the northwest coast of Ireland (Hanson and Lonergan, 2012; see Figure 10.4). Until further genetic analysis (assessment of population structure in the region) is undertaken, it is recommended that a western Britain, Ireland and western France assessment unit be used for MSFD assessments. It should be noted, however, that grey seals off western Scotland may be at carrying capacity (see Section 10.5), whilst other seal colonies within the western Britain, Ireland and western France management area may be in quite a different situation (see Section 10.9.6).

Note that grey seals are also ranging further south. Grey seals have been sighted in French waters of the Bay of Biscay, from Brittany to the Spanish boarder. The individuals are mainly juveniles, often young of the year, which is consistent with the high degree of juvenile dispersal in the species (Cecile Vincent, pers. comm.). Additionally, young grey seals are increasingly appearing along the Galician coast (NW Spain) (Begoña Santos, pers. comm.).

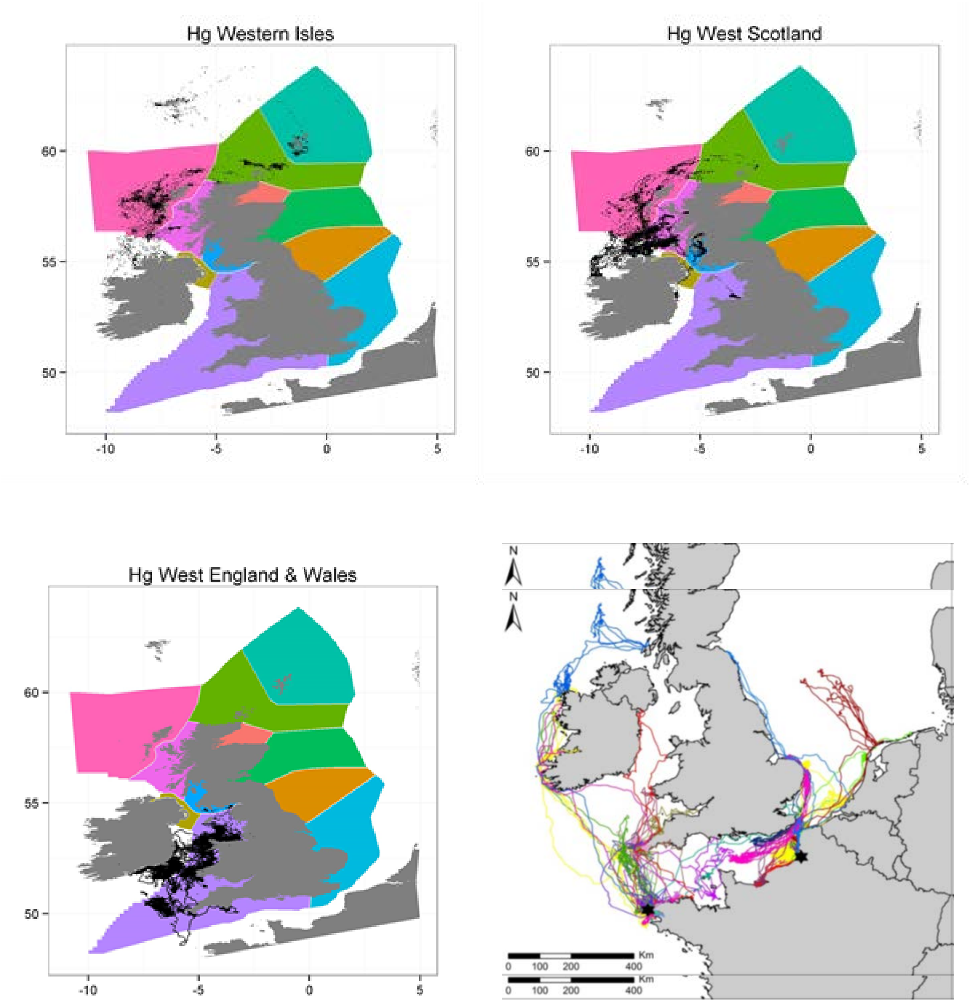


Figure 10.4. Top panels and bottom left panel: The distribution of telemetry points for harbour and grey seals tagged in each UK management unit between 1988 and 2012, taken from Hanson and Lonergan (2012).; bottom right panel: Tracks of grey seals from the French coasts of the western and eastern Channel (Université de La Rochelle / CNRS, Parc naturel marin d'Iroise, Océanopolis, Picardie Nature, Région Bretagne, Région Poi-tou-Charentes).

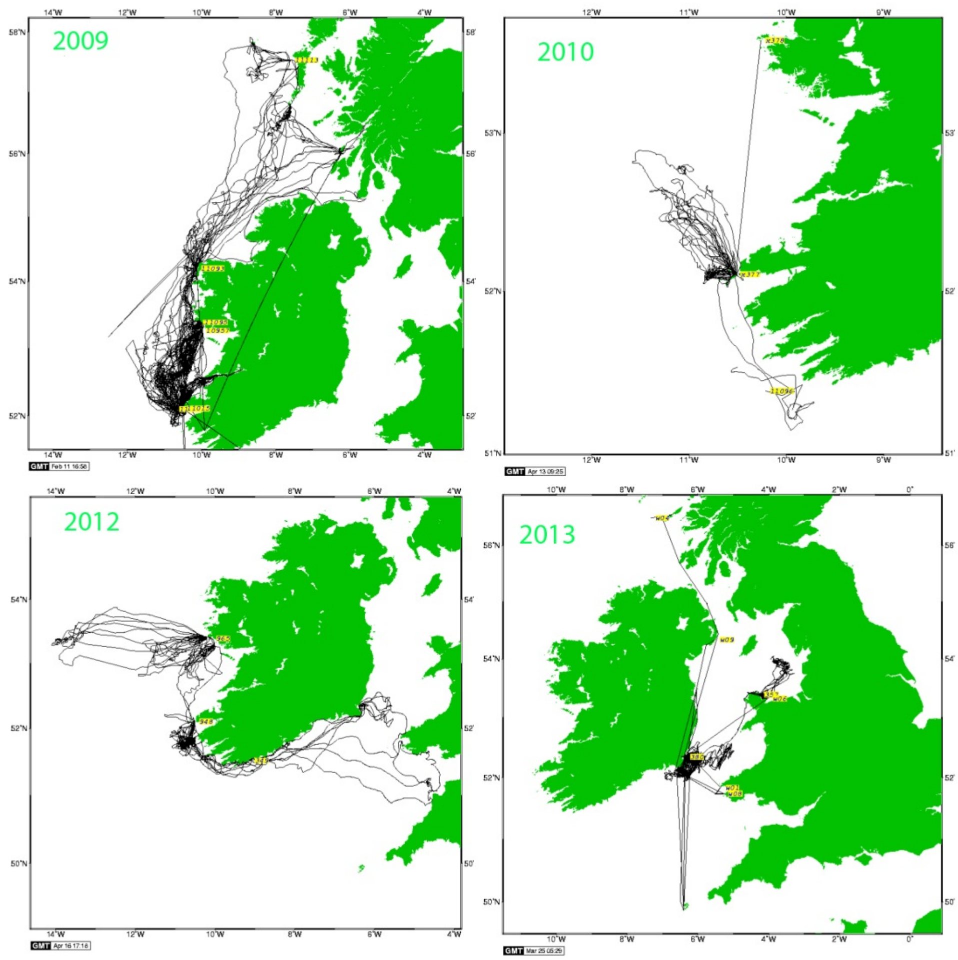


Figure 10.5. Tracks of grey seals tagged along the Irish coastline (Michelle Cronin unpublished data, CMRC, UCC).

10.1.3 Proposed Harbour Porpoise Assessment Units for OSPAR Regions II, III, IV

In 2013, the WGMME recommended the following AUs for harbour porpoise (*Phocoena phocoena*) delineated by ICES areas/division boundaries (except in one case; Figure 10.6):

1) North Sea (NS): Area IV, Divisions VIId and part of IIIa (Skagerrak and northern Kattegat), the boundary between NS and Kattegat/Belt Seas is currently being revised (Anders Galatius, pers. comm.);
2) Kattegat and Belt Seas (KBS): Part of Division IIIa (southern Kattegat) and Baltic Areas 22 and 23;
3) Western Scotland and Northern Ireland (WSNI): Divisions VIa, VIb2;
4) Celtic Sea and Irish Seas (CIS): Divisions VII with the exception of VIId;
5) Iberian Peninsula (IB): Divisions VIIIc and IXa.

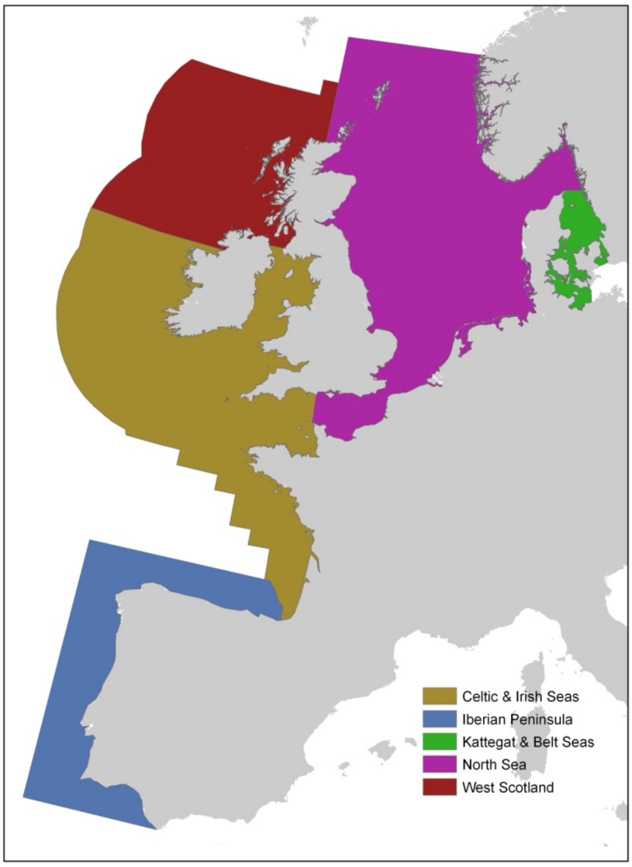


Figure 10.6. Harbour porpoise assessment units proposed for MSFD indicator assessments. The boundary of the North Sea AU to the west in Kattegat will be subject to change once the boundaries of the ASCOBANS conservation plan for harbour porpoise in the Western Baltic, the Belt Sea and the Kattegat have been fully decided.

10.1.4 Proposed Bottlenose Dolphin Assessment Units for OSPAR Regions II, III, IV

In 2013, the WGMME recommended the following AUs for bottlenose dolphins (*Tursiops truncatus*) (given from north to south; Figure 10.6).

Resident groups: Barra (Scotland; although for management purposes this group is included within the wider Scottish west coast group); Shannon Estuary (Ireland); Ile de Sein (France) Archipel de Molene (France); southern Galician Rias (NW Spain); Sado Estuary (Portugal).

Coastal groups: west of coast Scotland (UK); east coast of Scotland (UK); Irish Sea (Ireland and UK); Connemara–Mayo (northern and west coasts of Ireland); the English Channel/Celtic Sea (Ireland, UK and France); north coast of Spain; coast of Portugal (except for the Sado Estuary); the Azores (Portugal), Gulf of Cadiz (south coast of Spain) and Strait of Gibraltar (south coast of Spain).

Oceanic waters: a single AU for all continental shelf/slopes/oceanic waters outside 12 nm from the coast. It should be noted that although a separate AU is 'designated' for the North Sea (represented by ICES Area IV, excluding coastal east Scotland), there are very few bottlenose dolphin are seen in this area. Although there is no conclusive evidence, those seen are thought to belong to the East Scottish coastal group.

Updates to these AUs include, the re-naming of the Connemara–Mayo (northern and west coasts of Ireland) AU to the west coast Ireland AU. Although abundance estimates are only available for the Connemara-Mayo region, photo-id evidence suggests that these coastal animals range around the whole of the southwest/west coast from Youghal to Donegal (Simon Ingram, pers. comm).

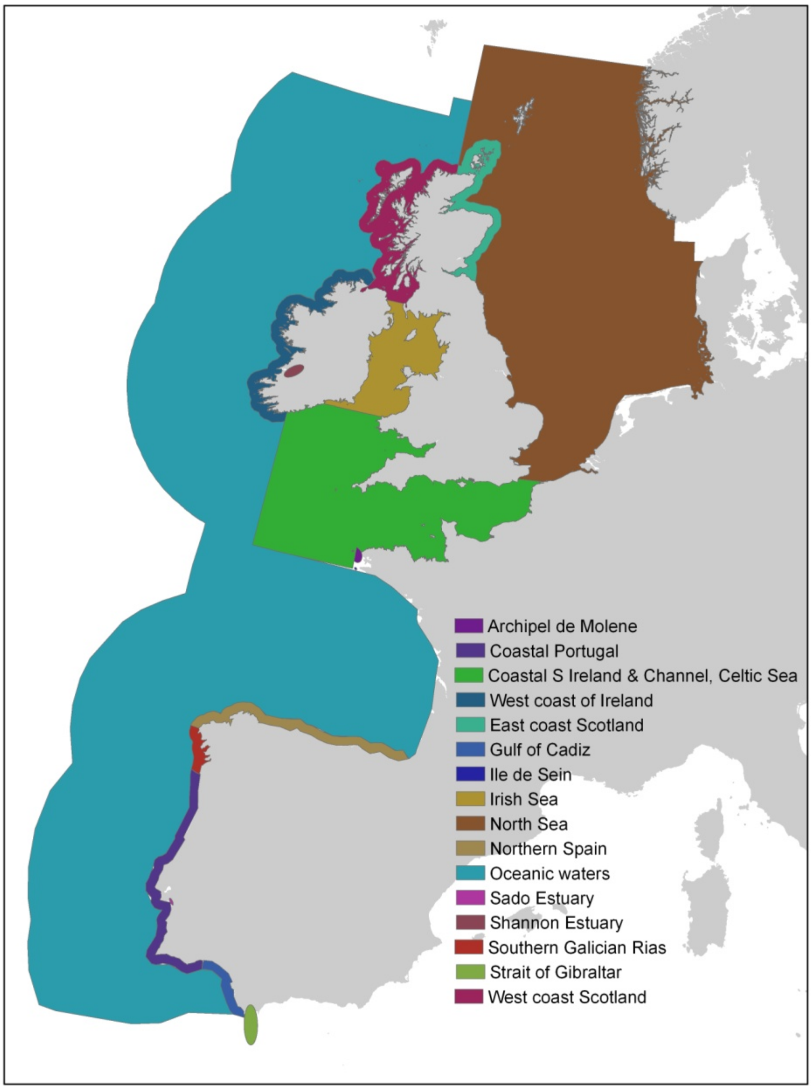


Figure 10.6. Bottlenose dolphin assessment units proposed for MSFD.

10.1.5 Proposed Common Dolphin Assessment Units for OSPAR Regions II, III, IV

WGMME (2013) concluded that only one population of short-beaked common dolphin (*Delphinus delphis*) exists in the Northeast Atlantic, ranging from waters off Scotland to Portugal, and there is thus a single AU.

10.1.6 Proposed White-beaked Dolphin Assessment Units for OSPAR Regions II, III, IV

WGMME (2013) recommended a single AU for white-beaked dolphin (*Lagenorhynchus albirostris*) around Britain and Ireland, comprising all relevant ICES areas and divisions. Additional AUs may be appropriate to northern Norwegian waters and waters around Iceland.

10.1.7 Proposed White-sided Dolphin Assessment Units for OSPAR Regions II, III, IV

WGMME (2013) recommended a single AU for white-sided dolphin (*Lagenorhynchus acutus*) in the eastern North Atlantic, comprising all relevant ICES areas and divisions.

10.1.8 Proposed Striped Dolphin Assessment Units for OSPAR Regions II, III, IV

Population structure in striped dolphins (*Stenella coeruleoalba*) in the NE Atlantic was reviewed by Murphy *et al*. (2007). Archer (1997) assessed osteological variation in striped dolphins. The main findings were a lack of detected differences between western and eastern Atlantic populations, but variations in skull size were identified between the eastern Atlantic Ocean and the Mediterranean Sea. On the whole, striped dolphins inhabiting the Mediterranean Sea have smaller sized skulls, compared to the NE Atlantic. It is believed that morphological homogeneity between eastern and western Atlantic striped dolphins is being maintained by genetic exchange across the North Atlantic Ocean (Archer, 1997). It was suggested that genetic exchanges might take place in the tropics, along the corridor formed by the Atlantic North Equatorial Current.

Direct sequencing of the mitochondrial genome from 57 skin samples from striped dolphins sampled in three populations (eastern Pacific, western Atlantic and Mediterranean) showed a high degree of haplotypic diversity (0.972). Average genetic distance between these populations was quite low (0.025), thus indicating a low degree of population divergence (Archer, 1996). Genetic analyses of mitochondrial DNA restriction enzymes (Garcia-Martinez *et al*., 1999) and analyses of nuclear microsatellite loci also revealed significant population subdivision between Mediterranean and adjacent Atlantic areas (Valsecchi *et al*., 2004; Bourret *et al*., 2007; Gaspari *et al*., 2007), indicating limited gene flow across the Strait of Gibraltar (Mirimin, 2007). No haplotypes were shared between the Mediterranean and the Northeast Atlantic samples (Garcia-Martinez *et al*., 1999). Both Valseechi *et al*., (2004) and Bourret *et al*., (2007) reported a higher level of allelic diversity in the Atlantic, compared to the Mediterranean population, with the Atlantic population being significantly more polymorphic than the Mediterranean population, in both nuclear and mtDNA. Bourret *et al*. (2007) identified a significant heterozygote deficiency in the Mediterranean Sea population, which may suggest significant inbreeding in the population, or due to the fact that samples were obtained from two or more reproductively distinct populations in the Mediterranean. Striped dolphins that died during the initial stages of the morbillivirus outbreak in the Mediterranean Sea (1990–1992) were significantly more inbred than those that died later (Valsecchi *et al*., 2004).

Genetic analyses using 13 microsatellite loci revealed no significant genetic differentiation between striped dolphins bycaught in the Celtic Sea and those that stranded on the southwest coast of Ireland, suggesting a lack of genetic differentiation due to movements between those two areas (Mirimin, 2007). Interestingly, levels of genetic diversity found in striped dolphins off the Irish coast were in the range of those found in the Mediterranean Sea in previous studies (Mirimin, 2007).

It appears that separate populations exist in the Northeast Atlantic and the Mediterranean Sea. There is however a lack of information regarding population structure in striped dolphins in the NE Atlantic. The WG recommends further genetic and morphological studies to be undertaken to investigate population structure in striped dolphins in this region. Until this work has been carried out, the WGMME **recommends** a single AU for striped dolphins within OSPAR Regions II, III, IV.

10.1.9 Proposed Minke Whale Assessment Units for OSPAR Regions II, III, IV

WGMME recommends a single AU for minke whale in the eastern North Atlantic, following designations of the International Whaling Commission (IWC), comprising all relevant ICES areas and divisions. The UK MU for this species has also followed a similar approach.

10.2 Quality and quality control in monitoring programmes for MSFD indicators

10.2.1 Generic issues

Successful monitoring programmes require clearly defined objectives, good design (based on power analysis) and well-articulated reference points/targets and indicators. In addition, there should be a well-defined mechanism to translate results into management actions to meet and policy objectives and a feedback mechanism to evaluate the success of the process.

Important principles of assessment and monitoring are taken up in the MSFD itself. In preparing assessments Member States need to ensure that the methodologies for assessing the environmental status of their marine waters are consistent across the marine region or subregion. Member States should establish coordinated monitoring programmes for the assessment, with monitoring programmes compatible within marine regions or subregions. Member states should (a.o.) endeavour to ensure that monitoring methods are consistent across the marine region or subregion so as to facilitate comparability of monitoring results.

Much of the current surveillance and monitoring of marine mammals in Europe will potentially contribute to MSFD monitoring programmes/indicator assessments. Although individual monitoring activities do not themselves constitute suitable monitoring programmes, for a variety of reasons including regional or local scope and, lack of continuity and/or funding, and (in some cases) poorly defined or inappropriate procedures.

In some cases, modification, integration and/or coordination of current monitoring activities could make them suitable to contribute to marine mammal MSFD indicator assessments for abundance and distribution (range and pattern). Possible actions include:

1) Extending geographic coverage and maximizing use of available data and samples, e.g. through integration of monitoring programmes run by NGOs.
2) Recognising that most marine mammal populations extend across MS boundaries, analysis of monitoring data should be undertaken at the most

appropriate spatial scales, e.g. at AU rather than MS or regional level, as far as allowed by relevant legislation.

3) Meeting conservation objectives, and especially considering points 1 and 2 above, implies adequate coordination and standardization of monitoring both within MS and between MS, ideally coupled with procedures for quality assurance and mechanisms to ensure adaptability. Setting up of common databases and sample banks may be appropriate.

4) Monitoring schemes should aim at using methodology that in a cost-efficiency context allows for the most powerful appropriate analyses to be performed on the obtained data and ensures comparability to other relevant programmes.

Last, but not least, prioritization may be needed. There is obviously a trade-off between the statistical power and cost of monitoring.

5) Not all proposed monitoring targets are realistically achievable and it is important to focus on those which can be achieved, while looking for alternatives to those which are not.

The current lack of standardization and coordination in some national marine mammal monitoring activities arises for various reasons, e.g. monitoring programmes have arisen to meet different objectives, are run by different kinds of organisation with differing levels of resourcing and training.

Developing universally applicable standard protocols is of course challenging. Partial standardization, e.g. guidelines for minimum data requirements to determine indicators for the metric of interest (e.g. abundance) and/or to allow datasets to be combined, may be more readily achievable. For example, the Joint Cetacean Protocol developed by the UK's JNCC is a database of effort-related cetacean visual survey data from a range of sources (Jewell *et al.*, 2012).

10.2.2 Power analysis for detection of trends

Monitoring schemes need to estimate parameters precisely and without bias in order for trends over space and time to be detected with confidence (ICES WGMME, 2010; 2011; 2012). It is essential that measurable objectives for monitoring are defined and, in particular, decisions need to be made with regard to acceptable levels and significance of change to be detected by the monitoring. Monitoring population abundance, for example, requires a decision about the size of change in a population that needs to be detected and how confident managers need to be about detecting that change. Confidence in detecting changes and trends can be gauged from power analyses.

For estimating population abundance, there is a close relationship between power, survey effort and the precision of the estimate determined by the monitoring. Studies have a high statistical power when they are very precise (i.e. small coefficient of variation, CV), the size effect is large (i.e. any change occurring accounts for a substantial proportion of the variation) and more lenient standards for determining significance (i.e. the α of a statistical test) are adopted. The precision of known estimates has implications for future monitoring requirements if targets, in relation to trend detection, are to be attained. As survey effort is increased, so the precision of estimates is increased (i.e. CV decreases) and consequently, the power to detect trends improves. Expending more effort at each sampling occasion or sampling more frequently, will increase survey effort over time with the result of improving CV.

If a monitoring scheme is unable to detect the required trends in the population with any degree of precision, this could have significant negative impact on the species. For example, if the monitoring put in place does not have the ability to detect a decline in the population until after 50% of the individuals have disappeared, it must be considered totally ineffective from a conservation perspective. In such a situation it is highly likely that any mitigation on the causes of the decline would be identified too late to be of any value to the population.

For almost all cetacean species, the cost of achieving monitoring with both a high level of power and also precision is high. A possible compromise, from a policy perspective, therefore needs to be made with a balance achieved between the power of the monitoring for each species to detect a change and the level of significance at which the trend is tested (i.e. α and β need to be as close as possible). This was discussed in detail in Section 8.4.5, WGMME (2008). The monitoring requirements and what can realistically be assessed will, therefore, vary between species depending on current knowledge.

WGMME (2008; 2010) proposed that monitoring should achieve ≥80% power and consideration given to the use of a significance level (α) of 0.2 rather than 0.05. Whilst the use of a significance level of 0.2 is unusual, it is considered to be a pragmatic approach to conservation. Using a significance level of 0.2, means that it could be concluded that a trend is occurring one in five occasions when it is not. Conversely, a power of 80% means that it could be concluded that no trend is occurring one in five occasions when it actually is. By fixing the power and significance to be equivalent, there is equal risk of an incorrect conclusion being drawn.

10.2.3 Cetaceans: abundance

The power to detect trends in abundance depends on the survey methodology, sample size, the statistical distribution of the parameter being measured and the magnitude of changes which must be detected. In relation to cetaceans, at least two general survey protocols need to be considered: (a) boat-based and aerial surveys for wide-ranging species and (b) assessments of coastal populations based on photo-ID and capture–mark–recapture, e.g. for resident bottlenose dolphin populations.

The difficulty in detecting cetacean population trends from surveys has been recognised at least since the early 1990s (see Forney *et al*., 1991; Edwards and Perkins, 1992). As discussed by Taylor and Gerrodette (1993), power to detect a decline in abundance decreases as populations become smaller, and can become unacceptably low in severely depleted populations such as that of the vaquita. Therefore, as they state, "*detection of a decline should not be a necessary criterion for enacting conservation measures for rare species*".

The magnitude of change in abundance to be detected and the time-scale of this change need to be clearly defined. For example, guidance on Habitats Directive FCS reporting issued by the European Commission (EC, 2011) has specified that monitoring should be able to "*detect a decline in abundance of more than 1% per year within a specific time period ....*". If the 'time period' is within the six year reporting cycle, then monitoring would need to have sufficient power (80%) to detect a decline of 1% per annum with statistical confidence (e.g. α = 0.05 or 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. '*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 nine 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 ten years) with the same CV is only 0.29*.'

However, a SCANS/CODA type survey repeated annually is not feasible either logistically or financially. WGMME, 2010 (see Section 6.4.2.1) proposed that a longer term view was required, that enabled consistent collection of data over time and where trends were only assessed once sufficient robust data were available. It was noted that for 'marine mammals, particularly cetaceans, this could mean trends are not reported with any degree of confidence for another ten to 15 years, maybe longer depending on the type of surveillance data collected and species under consideration.' Such an approach was adopted by the European Commission for the 2013 Favourable Conservation Status reporting under Article 17. Member States were required to provide an assessment of short-term (rolling 12-year time window) and longer term (24 years) trends.

Jewell *et al.* (2012) commented that measuring the effect of anthropogenic change on cetacean populations is hampered by our lack of understanding about population status and a lack of power in the available data to detect trends in abundance. Furthermore, often long-term data from repeated surveys are lacking, and alternative approaches to trend detection must be considered. However, they reported that increasing sample size by combining survey effort across a global scale did not necessarily result in sufficient power to detect trends because of high variability across surveys, species and oceans. Therefore, results from repeated dedicated surveys designed specifically for the species and geographical region of interest should be used to inform conservation and management.

The key MSFD indicators for cetaceans relate to abundance, distribution and bycatch. Although population demographic status is not specifically identified, information on population dynamics, health and causes of mortality can help us to interpret data on abundance, distribution and anthropogenic mortality. Indeed, in the absence of long-term baseline data, Huang (2013) recommended an integrative approach to estimate life-history and demographic parameters essential to status and risk assessment in cetacean populations. He proposes, where appropriate to the species concerned, a combination of line transect surveys, incorporating information on environmental characteristics to help identify critical habitat, use of capture–mark–recapture methods based on individual photo-ID data and use of stranded and bycaught animals to estimate life-history parameters, age-specific survivorships or mortality rates and population genetic diversity.

10.2.3.1 Abundance of wide ranging cetaceans

For most cetacean species in EU waters, SCANS and similar surveys currently represent the only reliable estimates of population size for the area. However such surveys tend to be decadal in frequency, limiting their statistical power to detect trends. Statistical power can be increased by carrying out more surveys, but these are costly and there is a law of diminishing returns. A combination of 10-yearly large-scale surveys and local/regional surveys with a higher (e.g. annual) frequency would significantly improve power to detect trends (e.g. the Joint Cetacean Protocol), although this is contingent on adoption of a standardized protocol for the local surveys. The power to detect trends through such a data collection mechanism is dependent upon the spatial and temporal nature of the data (Thomas, 2009; Paxton and Thomas, 2010). The absence of effort information is a major limitation to the value of much (but by no means all) opportunistic sightings data.

In addition to the large-scale decadal surveys, dedicated regional level surveys are required. Usually, as in the Netherlands (Geelhoed and Scheidat, 2013), the advised method to obtain robust unbiased abundance estimates is the use of line transect surveys applying distance sampling methodology. For Dutch North Sea waters, aerial surveys have been shown to be the most effective way to make use of the short time periods of good survey conditions. Representative coverage of the study area, through predetermined track lines, provides robust absolute density and abundance estimates. It should be noted that these dedicated surveys should be coordinated between MS on a regional seas basis, in order to cover a large range of species distribution/assessment unit, e.g. two large-scale surveys have been undertaken in the southern North Sea through the coordinated efforts of UK, the Netherlands, Germany, Belgium and Denmark (Gilles *et al.*, 2011; Geelhoed *et al.*, 2014).

The value of collection of environmental data during visual surveys is increasingly recognized, not only to allow survey effort to be standardised but to improve understanding of heterogeneity of distribution, facilitating habitat use modelling and improved abundance estimates (e.g. based on density surface modelling).

10.2.3.2 Examples of power in abundance surveys

Analyses have been carried out of the statistical power of large- and small-scale boat-based surveys, in both cases highlighting the point that only rather large changes in abundance would be detectable. For small-scale surveys, Thomas (2009) reported that very small trends in population abundance, such as 1% per year, are not detectable in any reasonable time-span. Trends in the order of 15–30% per year may be detectable over the six-year time span imposed by the EU Habitats Directive. Following this, Paxton and Thomas (2010) undertook power analysis using data in the Joint Cetacean Protocol database and all available data collected from the Irish Sea by both large- and small-scale surveys. Results showed that, for the harbour porpoise, bottlenose dolphin and common dolphin, quite small declines in modelled population density (0.3–2.2% per year) over a 6-year reporting period could be detected with a power of 0.8. For other species only very large changes in modelled population density would be detectable. As the modelled population densities relied on spatial and temporal smoothing, sudden declines would not necessarily be detectable. In addition, the method included variability due to observation error but ignored process error (random fluctuations in animal numbers from a smooth trend line; Paxton and Thomas, 2010). The results were also based on spatio-temporal models that may not be reliable (for further information see Section 6, ToR d).

In the United States, Taylor *et al*. (2007b) assessed scientists' ability to detect "precipitous" declines of marine mammal stocks (defined as a 50% decrease in abundance in 15 years) based on recent levels of survey effort. The percentage of precipitous declines that would *not* be detected as declines was 72% for large whales (n=23), 90% for beaked whales (n=11), 78% for small whales/dolphins/porpoises (n=69), 5% for pinnipeds counted on land (n=13), 100% for pinnipeds surveyed on ice (n=5), and 55% for polar bears/sea otters (n=6).

MacLeod *et al.* (2011) presented power analyses for large-scale abundance surveys. For harbour porpoise, the CV of the abundance estimate drops to almost 1/3 of its initial value as survey effort increases from 1000 km to 10 000 km. Power to detect changes between consecutive samples depends on the CV of the abundance metric, the duration of the monitoring period, the magnitude of change between samples and the significance level. Larger changes between consecutive samples can be detected with greater power for the same amount of survey effort. MacLeod *et al*. (2011) reported a power of around 0.8 to detect a 20% decline per year over a four year monitoring period comprising monthly one week boat-based surveys (see Figure 10.7).

Geelhoed and Scheidat (2013) noted that, in principle, the statistical power of the trend in harbour porpoise density depends on three factors: (1) the reliability of the yearly density estimates, (2) the magnitude of yearly fluctuations in harbour porpoise density and (3) the number of years in the time-series. As the statistical power is a complex interplay between these factors, they tested whether a trend in harbour porpoise density could be assessed under a previously estimated combination of coefficients of variation (CV) of within- and between-year densities. In addition, the highest porpoise densities in The Netherlands are seen in spring, and surveys in this period therefore have the highest power to detect trends. Monte-Carlo simulations showed that there was a decrease in the size of trend that could be detected after 12 years of surveys when comparing annual, biennial and triennial surveys (50, 58 and 65% decline detectable after 12 years). As yearly fluctuations in porpoise density could not be influenced by the monitoring programme, the easiest way to influence the reliability of trend detection is by changing the number of years in which the density of harbour porpoises is estimated. Halving the number of years in the programme (i.e. biennial surveys) would lead to a loss of 25% in statistical power: the smallest detectable annual trend falls from 6% to 7.5%. The main reason to increase the number of transect lines would be to increase the spatial coverage of the programme. In addition, the density estimates may become more reliable. The power of trend detection, however, is rather insensitive to this reliability (Geelhoed and Scheidat, 2013).

(a) (b)

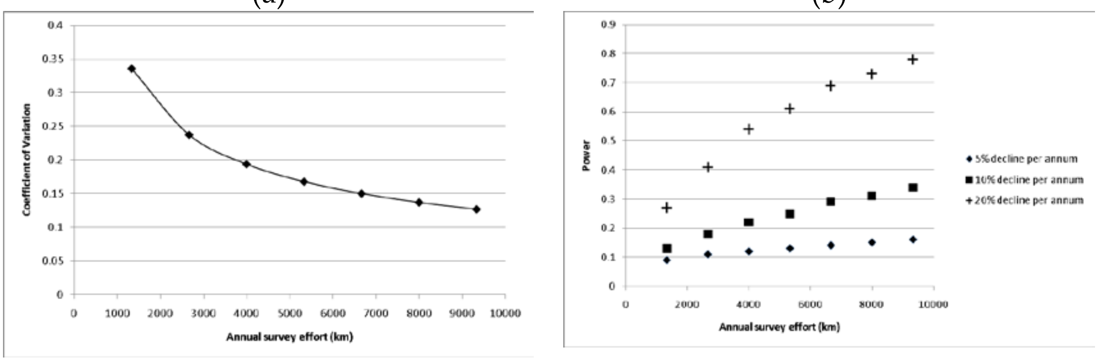


Figure 10.7. Power analysis for boat-based harbour porpoise monitoring surveys (from MacLeod *et al.*, 2011). An encounter rate of 0.02 harbour porpoise/km was used:

(a) Relationship between effort and total CV. CVs and effort were calculated for 1–7 days of survey effort per month for 12 months with six hours of effort per day at 10 knots.

(b) Relationship between power and effort for different levels of %change in the abundance per year. Power was calculated using TRENDS software (Gerrodette, 1993) for a four year monitoring period with annual monitoring and a one-tailed significance level (alpha) of 5%, assuming exponential decline and that CV was constant with abundance.

10.2.3.3 Assessing power for implementing conservation objectives for abundance

The Habitats Directive and the IUCN system provide explicit sets of criteria to evaluate the conservation status of species.

The reporting guidelines prepared by the Scientific Working Group of the Habitats Committee of DG Environment of the European Commission reported using an annual decline of 1% or more (during a 6-year reporting period) as a threshold value. Alternatively, another proposed threshold is more than 25% below favourable reference population. It is, however, for MS to decide what the favourable reference population of a particular species is for their waters. If the decline is larger, the conservation status is 'Unfavourable - Bad'. Whilst it is not possible for the levels of change prescribed under the Habitats Directive (≥1% per annum) to be detected, without excessive expense, Member States can come close with an appropriate level of monitoring (e.g. Paxton and Thomas, 2010). Clearly these criteria are applicable if only "favourable reference population" abundance can be specified. A possible example of this is provided by the SCANS II abundance estimates (Hammond *et al.*, 2013).

The IUCN Red List criteria define a species as 'vulnerable' when 'an observed, estimated, inferred, projected or suspected population size reduction of ≥30% over any ten year or three generation period, whichever is longer (up to a maximum of 100 years in the future), where the time period must include both the past and the future, AND where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible' (IUCN, 2013). For all cetaceans in the ICES area, three generations is >10 years. Therefore, using the IUCN example, the objective for monitoring abundance could be to detect a 30% decline over three generations for a particular species. The IUCN define a generation as 'the average age of parents of the current cohort (i.e. newborn individuals) in the population. Generation length, therefore, reflects the turnover rate of breeding individuals in a population. Generation length is greater than the age at first breeding and less than the age of the oldest breeding individual, except in taxa that breed only once. Where generation length varies under threat, the more natural, i.e. pre-disturbance, generation length should be used.'

According to Taylor *et al.* (2007a), the generation time for harbour porpoise is 11.9 years, whilst for other odontocete species on the shelf it averages at 19.6 years. Generation time is much longer for the larger species that generally reside off the continental shelf. Using these figures and assuming that the rate of abundance change is constant, and that the amount of change per annum decreases linearly (though declines will occur exponentially), a 30% decline over three generations for harbour porpoise equates to slightly less than 1% per annum over 36 years and approximately 0.5% per year for other odontocetes. However, true generation time for porpoises in Europe is probably much lower. In Scotland, the average age of mature females is approx. 7.5 years (Pierce, unpubl. data for Scotland). This may slightly underestimate generation time, since the youngest mature females will not produce a calf immediately, although there is also an opposite bias because ages were rounded down to the nearest whole year. Thus, 7.5 years is a more realistic figure for generation time of European porpoises and implies a need to detect a decline of 30% over 22.5 years, equating to approximately 1.5% per year.

If we consider the current decadal time-scale of the SCANS surveys and following figures in Taylor *et al*. (2007), which for at least some species overestimate generation time, the power to determine a 30% decline over three generations for all of the species is poor (<50% for all species except harbour porpoise which is 57%, see Table 2). Therefore, the frequency of such surveys needs to be increased if the defined trend is to be detected with good power. Using the CV of measured abundance estimates derived from these decadal surveys, it is possible to detect declines in harbour porpoise with a high degree of confidence if a large-scale survey were undertaken every five years (Table 10.2). Increasing the frequency to every third year, means that the detection of trends in species such as minke whale, common dolphin and white-beaked dolphin also become more viable, although with a slightly lower power (58–77%). An improved CV would also increase power to detect trends. If such surveys were extended off the continental shelf, then the detection of declines in pilot whales and sperm whales also become viable (63% power). However, applying Scottish data for porpoises reduces the generation time (thus shortening the assessment period) and implies that statistical power to detect a 30% change in three generations will be considerably lower/surveys will then have to occur more frequently to achieve a higher power.

Table 10.2. Precision (CV) of estimates of abundance and power (%) to detect a 30% decline in three generations (based on Taylor *et al.*, 2007) obtained from existing large-scale, decadal distance sampling surveys using ships and aircraft. Power is shown for both significance levels of 0.05 and 0.2. Cells coloured green where >80% power achieved, amber where >60% power achieved and red where the power is below 60%.

Monitoring activity	Species	CV of measured estimate of abundance	Power (%) to detect trends in abundance with survey every 10 years		Power (%) to detect trends in abundance with every 5 years		Power (%) to detect trends in abundance with every 3 years	
			α =0.05	α =0.2	α =0.05	α =0.2	α =0.05	α =0.2
SCANS (ships and aircraft)	Harbour porpoise (11.9 years generation time)	0.14	20	57	50	81	69	91
	Harbour porpoise (7.5 year generation time)	0.14	11	42	28	66	50	81
	White-beaked dolphin	0.3	12	36	20	46	30	58
	Minke whale	0.24	18	47	35	64	51	77*
SCANS-II (ships and aircraft)	Harbour porpoise (11.9 years generation time)	0.20	13	42	28	59	57	72
	Harbour porpoise (7.5 year generation time)	0.20	8	32	17	47	28	59**
	Short-beaked common dolphin	0.23	14	41	30	54	38	68*
	White-beaked dolphin	0.30	12	36	20	46	30	58
	Minke whale	0.35	11	34	18	43	28	55

***For minke whale and common dolphins, surveys every third year with a CV = 0.22 will achieve 80% power with an alpha of 0.2.**

**** For harbour porpoises (7.5 years generation time), biennial surveys with a CV=0.20, will achieve power 68% with alpha = 0.2. For biennial surveys to achieve a power of 80% with an alpha of 0.2, the threshold CV is 0.16.**

10.2.3.4 Abundance of coastal/resident cetacean populations

Photo-identification studies are recommended as the most appropriate monitoring methods for assessing abundance in small local coastal/resident populations. Wilson *et al.* (1999) undertook a power analysis for the Moray Firth bottlenose dolphin population (now described as the east coast of Scotland coastal group) to investigate the duration of monitoring program required to detect changes in population abundance at a 90% level of certainty. The population was estimated to consist of 130 animals (CV = 0.15) based on capture-mark-recapture analysis of photo-identified individuals. The authors showed that detection of a trend could only occur following more than eight years of monitoring. For the same population, Thompson *et al.* (2000) estimated that eleven years of annual surveys (CV=0.15) would be needed to detect a decline of

5% per year, whereas a decline of 1% per year would not be detected for over 30 years (Figure 10.8).

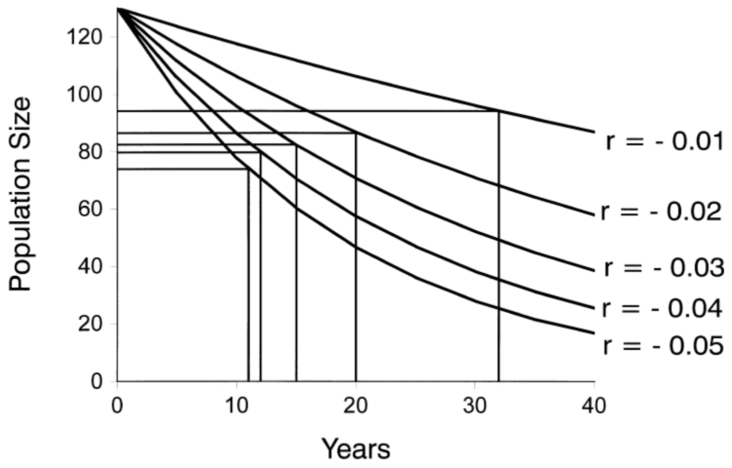


Figure 10.8. Variation in the time it takes to detect different rates of decline of the bottlenose dolphin population in the Moray Firth assuming that capture–recapture estimates of population size are made annually. The r is the annual rate of population change (From Thompson *et al.*, 2000).

Thompson *et al.* (2004) relaxed the power and significance to 0.1 (or 90%) and concluded that surveys should be conducted at least every three years to detect an annual decline of 5% within 12 years. However, annual summer surveys are still carried out between May and September for this population (Cheney *et al.*, 2012) and have an estimated CV of 0.15 (Wilson *et al.*, 1999). Such annual surveys have 100% power to detect a 30% decline over three generations (Table 10.3). Similarly, photo-identification surveys of the bottlenose dolphins in Cardigan Bay 2001–2011 have generated precise estimates of abundance (CV = 0.095; (Veneruso and Evans, 2012) which allow trends in abundance to be determined with 100% power (Table 10.3). Triennial surveys of these populations, which achieve a CV 0.16 (α = 0.05) or CV =0.22 (α = 0.2), can detect trends with 80% power or greater. However, these photo-identification studies provide considerably more information than just abundance estimates. Annual surveys enable, for example, individuals to be followed, thus contributing to other aspects of conservation status reporting such as assessing reproduction, mortality and age-structure.

Estimates of abundance for Risso's dolphin are only available from photo-identification data at a local scale, off Bardsey Island, North Wales (de Boer, 2013). Surveys (mainly opportunistic in nature) have been carried out since 1997 and were undertaken annually between 1999 and 2007. However, abundance estimation was only possible by pooling data over ten years. Assuming the CV estimated from these data and that effectively abundance is estimated once every ten years, this work has insufficient power to detect a 30% decline over three generations (Table 10.3).

Table 10.3. Power of existing bottlenose dolphin and Risso's dolphin photo-Identification monitoring to detect a 30% decline in abundance in three generations. Cells coloured red show where trend cannot be detected with the desired power of ≥80%, whereas green indicates objective is being met.

Monitoring activity	Species	CV of estimate of abundance	Estimate frequency	**Power (%) to detect trends in abundance**	
				α =0.05	**α =0.2**
Moray Firth	Bottlenose dolphin (inshore)	0.15	Annual	100	100
Cardigan Bay	Bottlenose dolphin (inshore)	0.095	Annual	100	100
Bardsey Island	Risso's dolphin	0.24	~10 years	18	47

Cañadas and Sagarminaga (2006) carried out a generic power analysis for detecting trends in bottlenose dolphin population size. In general, low rates of change require a large number of years to be detected (Figure 10.9a). Similarly, Englund *et al.* (2007) used CV values obtained for population estimates for bottlenose dolphins in the lower Shannon river, as calculated during 1997, 2003 and 2006, in a power analysis to predict the time taken to detect different hypothetical rates of population change when following a triennial reporting strategy (Figure 10.9b).

Englund *et al*. (2007) also considered how long it would take to detect a 5% annual decline, depending on the periodicity of the monitoring cycle (Figure 10.10).

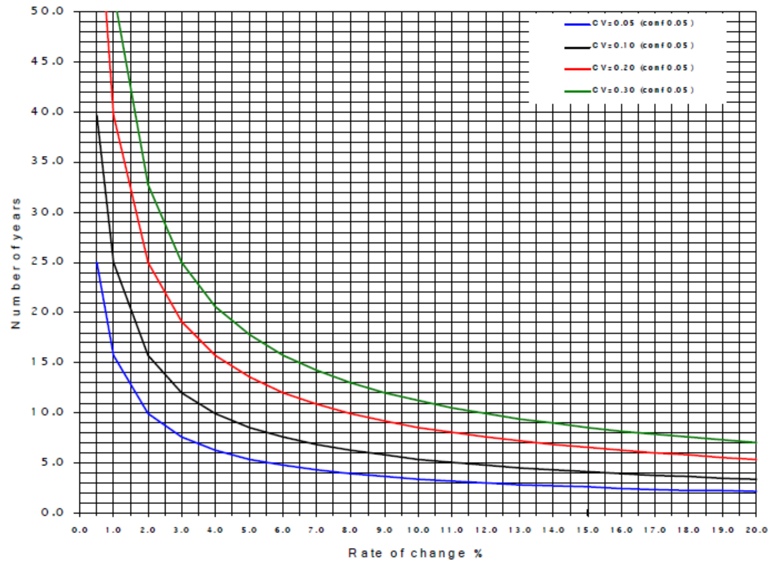
For a relatively large subpopulation of around 450 Indo-Pacific bottlenose dolphins (*Tursiops aduncus*), Ansmann *et al.* (2013) carried out a power analysis which suggested that to reliably detect a 5% decline in population would require >4 years of annual mark–recapture surveys, given the precision levels achieved.

10.2.4 Cetacean: distribution range and pattern

Abundance surveys are not well-suited to detect changes in distribution range, since detection of range changes requires most survey effort at the edges of the range, where density is lowest, and indeed, effort beyond the known range. Evidently, range expansion beyond the survey area will not be detected. However, both contraction of the range of a species within the survey area and changes in distribution within the range may be detected.

At least for those species whose distribution includes coastal areas, strandings offer an alternative data source to detect changes in distribution and range (e.g. (MacLeod *et al.*, 2005). In addition, models are being developed to link stranding locations to likely areas of origin (e.g. (Peltier *et al.*, 2012; Peltier *et al.*, 2013). However, most strandings and opportunistic sightings data tell us only about cetacean distribution in coastal waters.

Further discussions on the suitability of the indicator "distribution range and pattern" for marine mammals is addressed in Section 10.3.



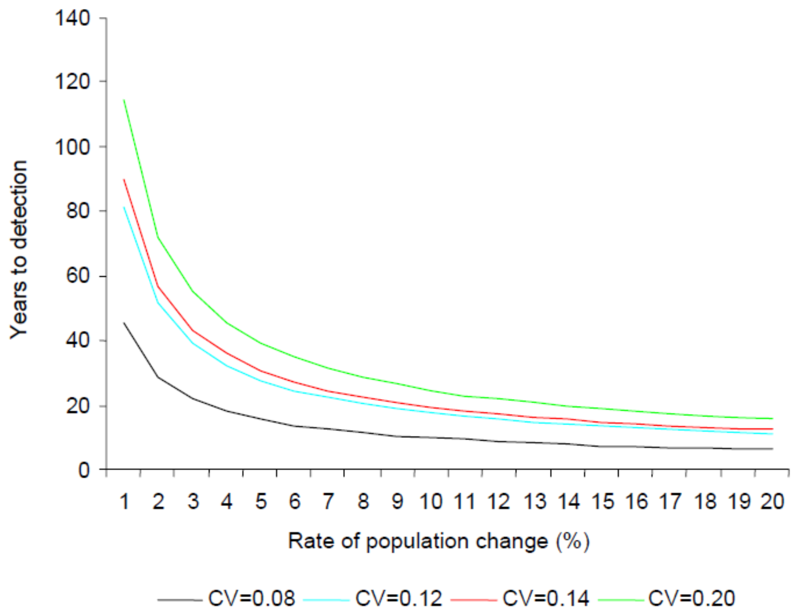


Figure 10.9. Power analysis results on the predicted time to detect different rates of change in the size of bottlenose dolphin population with different levels of precision for population estimates (four levels of CV) Detection probability is set at p=0.05. Andalucía and Murcia (from Cañadas and Sagarminaga (2006); Shannon river (Englund *et al.*, 2007)).

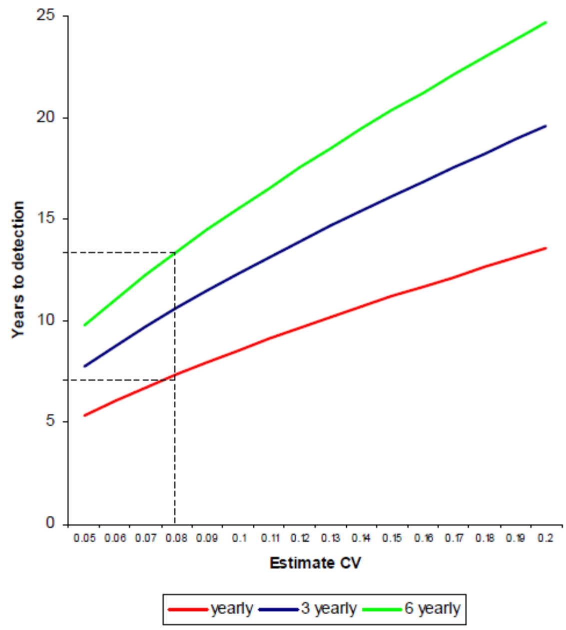


Figure10.10. Predicted time required to detect an annual rate of change of 5% in the Shannon dolphin population using three different monitoring periods. Dashed lines show the time to detection with CV=0.08, under three monitoring strategies. A rate of population change of 5% per annum would be detected after 12 years given a three year reporting cycle (Taken from (Englund *et al.*, 2007).

10.2.5 Seals: abundance

ICES WGMME (2012) summarised information on power analysis for monitoring of seals. Monitoring of seals is site-based, mainly using breeding colonies of grey seals and haul-out sites of harbour seals. A power analysis of Wadden Sea harbour seal data was used to assess the effectiveness of the existing survey schemes relative to the specific EcoQO. Aerial surveys during the moulting season did not meet the OSPAR guidelines (80% power and 5% probability to detect a change in abundance of minimally 10% over ten years) (Meesters *et al.*, 2007). The harbour seal monitoring programme in the Wadden Sea had sufficient power (80%) to detect a minimal trend of 2.2 % per annum in 10 years and 6% per annum in six years, as long as the variance within years was stable around the mean value. If the within-year variance increases, the power to detect trends may decrease rapidly. An analysis for the UK surveys that does not directly address these OSPAR guidelines, but assesses rates of change, is also available, which showed, for example, that the population in Orkney and Shetland declined by 40% (95% CI: 30–50%) between 2001 and 2006 (Lonergan *et al.*, 2007).

Data from surveys of harbours seals in southern Scandinavia waters were assessed against requirements of the EU Habitats Directive (1% decline, with a 6-year reporting period). At a power of 0.8, it was only possible to detect an annual change of 10% in abundance after six years with a 20% significance level, and after eight years with a 5% significance level. It was observed that higher power to detect changes in abundance could be achieved within a 6-year period in some areas, whereas a longer time-series is required in other areas, due to within-season and between-year variances (Teilmann *et al.*, 2010). Overall, power was typically doubled when carrying out annual surveys compared with every second year, and the power increased substantially when carrying out replicate surveys during the annual moult. The gain in power increases steeply up to three annual replicates (Teilmann *et al.*, 2010). This work again emphases the need to take a longer term (e.g. three generations) approach to assessing trends.

Power analysis has not been undertaken on any of the grey seal EcoQO subunits to assess the effectiveness of the existing survey schemes relative to the specific EcoQO, which was a recommendation of the ICES WGMME in 2009.

10.3 Baselines and target setting for marine mammals

Through the work of ICG-COBAM, OSPAR has produced an advice manual for Contracting Parties on the development of appropriate indicators and targets for determining Good Environmental Status (GES) in biodiversity (ICG-COBAM, 2012). This manual outlined the main approaches for setting baselines and targets for all elements of biodiversiy, including marine mammals (Box 10.1).

BOX 10.1.

APPROACHES TO SETTING BASELINES ARE:

METHOD A (REFERENCE STATE/NEGLIGIBLE IMPACTS) - BASELINES CAN BE SET AS A STATE IN WHICH THE ANTHROPOGENIC INFLUENCES ON SPECIES AND HABITATS ARE CONSIDERED TO BE NEGLIGIBLE;

METHOD B (PAST STATE) - BASELINES CAN BE SET AS A STATE IN THE PAST, BASED ON A TIME-SERIES DATASET FOR A SPECIFIC SPECIES OR HABITAT, SELECTING THE PERIOD IN THE DATASET WHICH IS CONSIDERED TO REFLECT LEAST IMPACTED CONDITIONS;

METHOD C (CURRENT STATE) - THE DATE OF INTRODUCTION OF AN ENVIRONMENTAL DIRECTIVE OR POLICY CAN BE USED AS THE BASELINE STATE. AS THIS MAY REPRESENT AN ALREADY DETERIORATED STATE OF BIODIVERSITY, THE ASSOCIATED TARGET TYPICALLY INCLUDES AN EXPRESSION OF NO FURTHER DETERIORATION FROM THIS STATE.

APPROACHES TO TARGET-SETTING ARE:

METHOD 1. DIRECTIONAL OR TREND-BASED TARGETS

i. DIRECTION AND RATE OF CHANGE

ii. DIRECTION OF CHANGE ONLY

METHOD 2. TARGETS SET AT A BASELINE

METHOD 3. TARGET SET AS A DEVIATION FROM A BASELINE

For marine mammals, ICG-COBAM (2012) specifically noted that 'taking into account limited data availability for cetaceans, method 1 is advised for target setting, while any of the approaches to set a baseline (methods A, B and C) could be applicable, depending on data and the history of hunting.' Also noting that 'target-setting method 1 and baseline-setting method C are advised, building on experience with EcoQOs.' In addition, expert judgement can be used to supplement these approaches to setting baselines and targets, thereby allowing a range of disparate information to be brought together.

In general, ICG-COBAM (2012) proposed that this would lead to the adoption of an approach similar to that of the Habitats Directive for determining Favourable Conservation Status but with assessment units based on biological populations (rather than Member State political boundaries). It was also proposed that 'where historic data indicate population size, distribution and condition were greater in the past, GES targets should seek a clear improvement in these criteria (rather than simply maintaining them at current state).'

In the absence of any reliable information from which to derive baseline and target states, ICG-COBAM (2012) notes that an alternative approach based on the impact of particular pressures could be adopted.

For seals, ICG-COBAM (2012) proposes the use of the existing EcoQOs, on harbour seal population size and on grey seal pup production (a proxy for breeding population size) as targets. 'Both EcoQOs use a current baseline of a five-year running mean (Baseline-setting method C) and a directional/ trend based target (rate of change) (Target-setting method 1): taking into account natural population dynamics and trends, there should be no decline of ≥10% within any of eleven subunits (re. harbour seal) or nine subunits (re. grey seal) of the North Sea.' However, the EcoQOs were designed to alert Contracting Parties to OSPAR that all is not necessarily well with an important part of the North Sea's mammal fauna. If the EcoQO was not met, then this should trigger research into the causes of the change rather than result in management action. Consequently, the EcoQO may not necessarily indicate whether GES has been achieved. ICG-COBAM (2012) noted 'the use of a current baseline may not be appropriate in the context of GES because it does not indicate what the aspirations for seal populations should be. Secondly, the 10% target may also not be appropriate to GES, given that it was not developed to be a statutory threshold: 10% was the level ...... at which 'social concern' is usually raised.'

10.3.1 Approaches for other mobile species

The range target set by the ICG-COBAM expert bird group is 'No major shifts or shrinkage in the range of marine birds in 75% of species monitored (separate assessments for each functional group, and for range of breeding birds and range of inshore water birds)' with an associated baseline of 'Set as past distributions where data are available; otherwise use the start of new time-series'**.** Similarly, the abundance target is 'Changes in abundance of marine birds should be within individual target levels in 75% of species monitored (separate assessments for each functional group, and for breeding and non-breeding aggregations). Species-specific annual breeding abundance should be more than x% and less than y% of the baseline (values of x and y can be species-specific)' with the baseline 'Set as past distributions where data are available; otherwise use the start of new time-series'.

The UK expert fish group took a slightly different approach, proposing a method for setting indicator-level targets for the number of species-specific metrics required to meet their trends-based metric-level targets. This is based on demonstrating significant departures from the binomial distribution as a mechanism to set objective species-level targets using upper- and lower- percentile thresholds for sensitive species (Greenstreet *et al.*, 2012).

Due to the relatively small number of cetacean and seals species by comparison to bird and fish species groups covered by MSFD, neither of these approaches are considered feasible for marine mammals.

10.3.2 Targets and baselines for MSFD mammal indicators

Targets need to be set in relation to reference levels and conservation objectives, while recognising the limits of statistical power to detect change based on logistically feasible monitoring. The issues are whether we can identify changes beyond the normal range of natural variability, and what amount of change is realistically detectable. As outlined in Section 10.2, power to detect trends may be increased by good survey design, and by incorporating multiple data sources, which may also improve understanding of any changes. A key issue is to avoid setting impossible targets.

Therefore, the WG **recommends** that power analysis be undertaken on all proposed national and regional seas monitoring programmes, to identify the power of existing and future monitoring activities to detect rates of change/trends. As an example, Danish harbour seals are monitored in three different management areas, the Limfjord, Kattegat and the western Baltic, and the seals are counted three times during the moulting season. The power to detect changes is different between the areas, because the within-season and between-year variances of the counts are different in the three areas. In the area with the least variance, a time-series of 10 annual monitoring seasons yields a power of 0.80 to detect an annual change in abundance of 2–3%. In the area with the highest variances, ten annual monitoring seasons yields a 0.80 power to detect an annual change of 10–11%. Increasing the number of annual surveys would only increase the power of detection minimally (Teilmann *et al.*, 2010).

Information yielded on power of current and future monitoring programmes should then inform target setting, though it should be noted that targets should be set based on both scientific and societal decisions. As outlined last year, proposed monitoring should achieve ≥80% power and consideration should be given to the use of a significance level (α) of 0.2 rather than 0.05.

10.3.3 Cetaceans: targets

10.3.3.1 M-2 “distributional range and pattern of cetaceans species regularly present”

For M-2 “distributional range and pattern of cetaceans species regularly present”, the target proposed is “Maintain populations in a healthy state, with no decrease in population distribution with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state”.

The distributional range and pattern of a species is the geographical area where it is located.

IUCN (International Union for the Conservation of Nature) uses geographic range in the form of either “extent of occurrence” or “area of occupancy” or both to determine the conservation status of a species[2]. Extent of occurrence is defined as “the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred or projected sites of present occurrence of a taxon, excluding cases of vagrancy”. This metric is obtained by drawing a polygon that contains all sites of known occurrence and should be the smallest polygon in which no internal angle exceeds 180 degrees. Area of occupancy is defined as “the area within its 'extent of occurrence' which is occupied by a taxon, excluding cases of vagrancy”. Because the size of this area is a function of the scale at which is measured, it is recommended that this scale should be “appropriate to relevant biological aspects of the taxon, the nature of threats and the available data” (IUCN, 2013).

For highly mobile groups such as cetaceans however, the range of a species/ population is sometimes difficult to determine and quantify accurately. There are reports of sightings of individuals outside what was considered the main range of a species, but

[2] http://www.iucnredlist.org/technical-documents/categories-and-criteria/2001-categories-criteria

when does this constitute a range expansion or an excursion beyond the range? (while the IUCN specifically acknowledges the concept of "vagrancy" it seems not to be well-defined). In addition, some cetacean species have well defined seasonal migrations. Detecting a range extension (animals appear where previously there had been no sightings) is relatively simple (except for the aforementioned issue of distinguishing vagrancy from range extension) if surveying outside the known range of a species but the determination that a species has disappeared from an area is much more difficult to establish unambiguously.

The IUCN sets criteria to determine the conservation status of a species between critically endangered, endangered and vulnerable, based on the extension of its geographical range (measured as extent of occurrence or area of occupancy), on the state of its range (if it is severely fragmented for example) and/or on its evolution in time (as observed, inferred or projected continuing decline or extreme fluctuations). "Severely fragmented" is used to define those situations in which "increased extinction risk to the taxon results from the fact that most of its individuals are found in small and relatively isolated subpopulations". The criteria further specify that in certain circumstances this may be inferred from habitat information.

The Habitats Directive (92/43/CEE) requires Member States to assess the Conservation Status of all cetacean species in relation to natural range, in addition to population size, habitat (extent and condition) and future prospects. Range was defined as "the outer limits of the overall area in which a habitat or species is found at present. It can be considered as an envelope within which areas actually occupied occur as in many cases not all the range will actually be occupied by the species or habitat".

The guidelines for the Habitats Directive report on criteria for determining the status of a species range is provided in the EC´s general evaluation matrix (Annex C)[3] and shown in Table 10.4.

The 'favourable reference range' is defined as the "range within which all significant ecological variations of the habitat/species are included for a given biogeographical region and which is sufficiently large to allow the long-term survival of the habitat/species". In addition, it is stated that the "favourable reference value must be at least the range (in size and configuration) when the Directive came into force; if the range was insufficient to support a favourable status the reference for favourable range should take account of that and should be larger (in such a case information on historic distribution may be found useful when defining the favourable reference range); 'best expert judgement' may be used to define it in absence of other data".

[3] https://circabc.europa.eu/sd/a/5c427756–166d-4cc8-a654-fca8bfae3968/Art17%20-%20Reporting-Formats%20-%20final.pdf

Table 10.4. Assessing conservation status of a species.

Parameter	**Conservation Status**			
	Favourable ('green')	**Unfavourable- Inadequate ('amber')**	**Unfavourable- Bad ('red')**	**Unknown (insufficient information to make an assessment)**
Range	Stable (loss and expansion in balance) or increasing And Not smaller than 'favourable reference range'	Any other combination	Large decline: equivalent to a loss of more than 1% per year within period specified by MS Or More than 25% below favourable reference population	Large decline: equivalent to a loss of more than 1% per year within period specified by MS Or more than 10% below favourable reference range
Population	Population(s) 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

In reporting cetacean species status under Article 17 of the Habitats Directive, many MS took the "*favourable reference range*" as the extent of their shelf waters, since the exact distribution of these highly mobile species is difficult to pinpoint in space and time. In fact, in 2007, most Member States submitted distribution rather than range data for favourable conservations status assessments, which was recognized by ETC/BD in many of the collated assessments (WGMME, 2009). Because MS reported differently under range (and other parameters used in the assessment), new revised guidelines and reporting formats were introduced to provide better coherence for the second reporting period. In these revised guidelines, a specific mention is provided for range. It is stated that "Range is a technical parameter allowing for assessing the extent and the changes in the habitat type or species distribution". In addition guidelines are provided on how to calculate it: "The range should be calculated based on the map of the actual distribution using a standardised algorithm".

A surface area can be drawn using the sightings of the species and this surface area can be evaluated against the favourable reference range and/or against time to determine trends. Whether this approach provides meaningful information on the conservation status of cetacean species depends on the quality of the data since incomplete coverage, particularly of sparsely distributed species, could lead to wide changes in surface area. The Directive has set specific targets of range decline to assess conservation status (see Table 10.4).

Using sightings with associated effort obtained from systematic and non-systematic surveys carried out in UK and Irish waters, an attempt has been made to analyse trends in both distribution and abundance. For this initiative, initially called the Joint Cetacean Database and now the Joint Cetacean Protocol, all sightings obtained by different organisations were put together into a single database. The analysis of these data is still ongoing. A preliminary analysis of the data was conducted to determine the level of change in range that could be detected over the six year reporting period imposed by the Habitats Directive (Thomas, 2009).

Because of the difficulties in accurately assessing declines in range following the experiences of MS with the assessment and reporting under Article 17 of the Habitats Directive, the usual methods of determining targets and baselines are deemed inappropriate to M-2. Therefore, the WG recommends that, because it is not possible to propose a firm and measurable baseline, metric and target for the current M-2 indicator, no separate monitoring be performed for M-2 and this indicator be removed from the list of common OSPAR indicators. The WG agreed that distribution changes should act as warning signals and research should be carried out to investigate the causes of those changes, especially to determine if they have an anthropogenic cause. This indicator should therefore be subsumed within M-4-Cetacean abundance indicator, and monitoring of a species′ range or its pattern of distribution within its range should be carried out as part of the monitoring to determine population abundance and its evolution over time (see next section).

10.3.3.2 M-4 "abundance at the relevant temporal scale of cetacean species regularly present"

For OSPAR mammal common indicator M-4 "abundance at the relevant temporal scale of cetacean species regularly present", the target proposed is "Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state".

Abundance is one of the most important parameters when considering the ecological role of a species and perhaps the most important when trying to assess the status of a population as negative trends should be avoided.

In some cases, absolute abundance estimates are not available and indices of abundance have been developed to determine trends in population abundance (Evans and Hammond, 2004). Although these indices can be used to detect trends (at least if adequate measures of uncertainty are available), estimates of absolute abundance are still needed to assess the possible impact that anthropogenic threats are having on the population and to define reference limits.

As outlined earlier, the IUCN uses levels of population decline to establish the conservation status of a species. These levels are determined either by observation, inference or if the decline/population size reduction is suspected over a time period of the last ten years or three generations, whichever is longer. Levels are variable, contingent on the Red List category, and dependent on whether the causes of the decline are known, reversible and have stopped (higher level) or not (lower level). It should be noted that the term 'population' is used in a specific sense in the Red List Criteria that is different from its common biological usage. Population is defined as the total number of individuals of the taxon. For functional reasons, primarily owing to differences between life forms, population size is measured as numbers of mature individuals only (IUCN, 2013).

Estimates of population abundance and trends over time are required as part of the assessment of FCS for the Habitats Directive. Determining the conservation status of a species must follow the criteria provided by the EC´s general evaluation matrix (Annex C) reproduced in Table 10.4.

Although the cetacean abundance indicator does not specifically specify a “significant” decrease in population size with regard to the baseline, definitive targets need to be set. One criterion for unfavourable status in the Habitats Directive is a decline of more than 1% per year (during a six year reporting period). As outlined in Section 10.2, several exercises using power analysis have been undertaken to determine the power of the current monitoring programmes to detect the level of changes required by the Habitats Directive. In general, because of the logistic difficulties of surveying wide-ranging marine mammals, the estimates of absolute abundance that have been obtained for many populations have poor precision and therefore they have low power to detect trends in the short to medium term. Higher power was obtained for smaller coastal/resident populations that are surveyed more frequently.

As the target of detecting a 1% annual decline adopted by various policies is very difficult to achieve and due to the long generation times of cetacean species, the WG proposes taking the IUCN approach. The IUCN defines a species as “vulnerable” when there is a “population size reduction of ≥30% over any ten year or three generation period, whichever is longer (up to a maximum of 100 years in the future);” criteria for “endangered” and “critically endangered” were ≥50% and ≥80%, respectively. In the case of cetaceans, generation time is such that multiplying this value by three presents a longer time period than the stipulated ten years. Therefore the target becomes a reduction of 30% in population size over three generations. However, for certain species assessment units, with small abundance, i.e. coastal and resident bottlenose dolphins, it is important to identify biologically significant rates of decline on a shorter-time-scale than three generations. Thus, for the purposes of setting targets for M-4, the WG proposes “population size reduction of ≥30% over any ten year or three generation period, whichever is appropriate to the species concerned”. Generation time can be calculated based on life table methodology using life-history data available for several species / areas combinations (e.g. (Taylor *et al.*, 2007a). Population size is the total number of individuals.

Using these proposed criteria, the WG **recommends that large-scale surveys, such as SCANS and CODA be undertaken every six years, instead of decadal.** Although these surveys may not have sufficient power for some of the more common species (see Table 10.2, Section 10.2.3.3), there are cost constraints and resource limitations for undertaking such large-scale surveys within European waters. For photo-id studies, power analysis should be undertaken for those monitoring programmes not assessed to date, and survey frequency adjusted accordingly.

10.3.4 Cetacean; baselines

As noted by ICG-COBAM "although the most robust way to set baselines for marine mammals is based on historical data, these are not available at the appropriate spatial and temporal scale. Moreover, the historical abundance of many cetacean species (i.e. pre-commercial hunting) is unknown and cannot realistically be restored (where it is known to have declined) as today's marine environment is very different"[4]. ICG-COBAM proposed setting baselines for the targets are the same as the baselines for the Habitats Directive (i.e. 1992 or the closest best estimate).

Table 5 presents the baselines recommended by the WG for cetacean species regularly present. For wide-ranging species such as harbour porpoises, common dolphins and minke whales, it is recommended to use abundance estimates from large-scale surveys such as SCANS I in 1994, SCANS II in 2005 and CODA in 2007, depending on which initial survey/s covered the majority of the species distribution range. Whereas, for smaller assessment units e.g. coastal and resident bottlenose dolphins, the first abundance estimates calculated using small-scale surveys or photo-id and mark–recapture are presented.

10.3.5 Seals: targets

10.3.5.1 M-3, "Abundance of grey and harbour seal at breeding and haul-out sites, respectively"

For M-3, "Abundance of grey and harbour seal at breeding and haul-out sites, respectively" the target was set to: "Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state", and for indicator For M-5 "Grey seal pup production", the target was set to "No statistically significant long-term average decline of ≥10% at each Management Unit". M-3 should apply to both grey and harbour seals, and M-5 should only apply to grey seals.

Detecting a ≥10% decline in grey seal pup production at each AU may not be realistically achievable and requires assessment. Power analysis should be undertaken on all current and future seal monitoring schemes relative to the specific MSFD indicator target. Additionally, it is essential that targets for both indicators are time-bound. HELCOM's core indicator for biodiversity 'population growth, abundance and distribution of marine mammals' uses primarily population growth rate for determining Good Environmental Status. It is proposed that "as long as the populations have not reached the carrying capacity of the environment, a rate close to the intrinsic growth rate, indicates GES. In the carrying capacity or near it, the GES is maintained when the populations do not decrease more than 10% over ten years" (Härkönen *et al.*,

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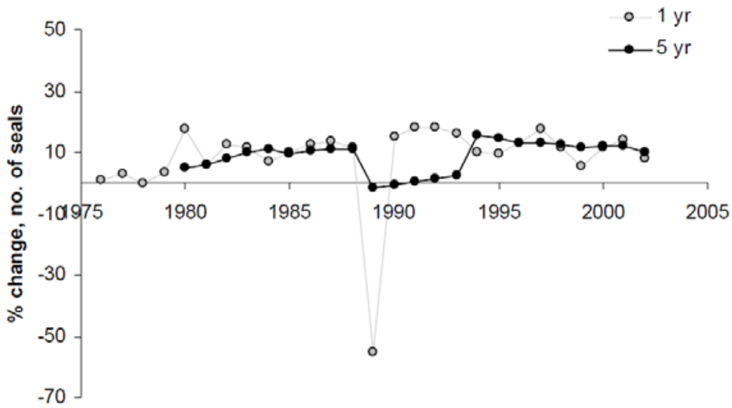
[4] ICG-COBAM advice manual – http://www.ospar.org/documents/dbase/publications/p00581_advice%20document%20d1_d2_d4_d6_biodiversity.pdf

2013). Currently, all populations in the Baltic Sea are assessed against their intrinsic growth rate (Härkönen *et al*., 2013).

In 2004, the WGMME reviewed the targets for seal EcoQOs, "Taking into account natural population dynamics and trends, there should be no decline in harbour seal population size (as measured by numbers hauled out) of ≥10% over a period of up to ten years"" and "Taking into account natural population dynamics and trends, there should be no decline in pup production of grey seals of ≥10% over a period of up to ten years"", which are similar to the proposed common seal MSFD indicators. WGMME (2004) reported that targets would be triggered rather often due to the interannual variation in numbers of seals (both pups counted or numbers on haul-outs). This level of "alarms" was felt by WGMME to be too high, and thus the WG suggested that a five-year running mean might be applied to these figures (see Figure 10.11). Such an approach would detect long-term changes in pup production or haul-out numbers for grey seals and harbour seals, respectively. The disadvantage of this is that mortality events, such as caused by epizootics, might not trigger the EcoQO. WGMME (2004) felt that this was not a major disadvantage as large mortality events appear to already be investigated in depth, whereas more subtle long-term changes might easily be overlooked. If the level of "false positive" was felt to be too high with a five-year running mean, it might be possible to switch to a three-year running mean (ICES WGMME, 2004). This approach is **recommended** again by the WG.

The EcoQOs were thus reformulated as: "taking into account natural population dynamics and trends, there should be no decline in harbour seal population size (as measured by numbers hauled out) of ≥10% as represented in a five-year running mean or point estimates (separated by up to five years) within any of eleven subunits of the North Sea" and "taking into account natural population dynamics and trends, there should be no decline in pup production of grey seals of ≥10% as represented in a five-year running mean or point estimates (separated by up to five years) within any of nine subunits of the North Sea" (OSPAR, 2010).



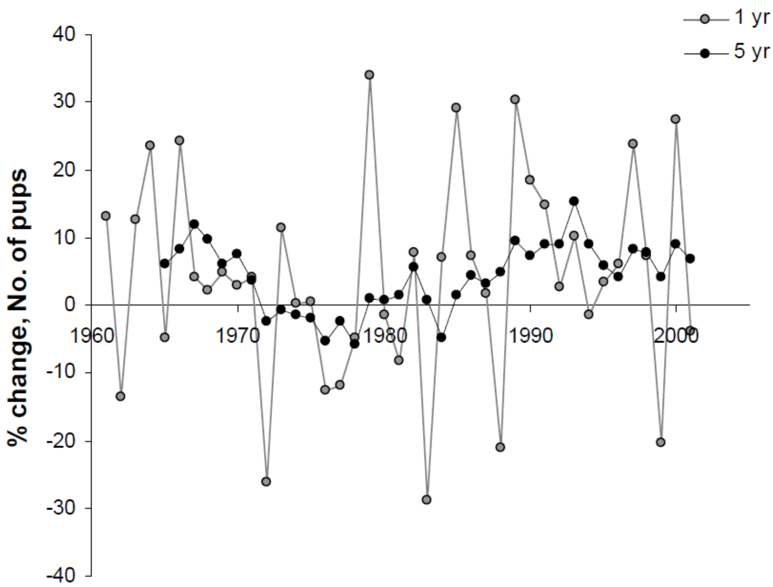


Figure 10.11. (A) Annual and five-year running means of changes in harbour seal counts in Niedersachsen and Schleswig Holstein (M. Scheidat, pers. comm.) (B) Time-series of annual and five-year running mean changes in estimated grey seal pup production at major UK breeding sites in the North Sea, except Helmsdale, Orkney, and Shetland (after (Duck, 2002) (Taken from ICES WGMME, 2004).

Table 10.5. Baselines proposed for cetacean species regularly present. Note that estimates encompass OSPAR regions II, III, IV and V.

Species	Assessment Units	Monitoring level	Reference	Baseline data collection, Survey	Baseline data collection Year	Abundance estimate	CV	SE	95% CI	Reference
Harbour Porpoise	Kattegat and Belt Seas	AU/subpopulation?	(Evans *et al.*, 2009)	SCANS I	1994	27,923	0.46			(Sveegaard *et al.*, 2013)
Harbour Porpoise	North Sea	AU	(ICES WGMME 2013)	SCANS I	1994	227,918	0.15		204,478-366,939	(Hammond *et al.*, 2002)
Harbour Porpoise	West Scotland and Northern Ireland	AU	(Evans *et al.*, 2009)	SCANS II	2005	21,462	0.42		9,740–47,289	(Macleod *et al.*, 2009, Hammond *et al.*, 2013)
Harbour Porpoise	Celtic and Irish Seas	AU	(ICES WGMME 2013)	SCANS II	2005	106,382	0.32		57,689-196,176	(Macleod *et al.*, 2009, Hammond *et al.*, 2013)
Harbour Porpoise	Iberian Peninsula	Population	(Fontaine *et al.*. 2007)	SCANS II	2005	4,398	0.92		948-20,410	Hammond *et al.*, 2013
Bottlenose Dolphin	North Sea	AU	(ICES WGMME 2013)	Very few BNDs are seen in this area and, although there is no conclusive evidence, those seen are thought to belong to the Coastal Scottish group.		none				
Bottlenose Dolphin	East coast of Scotland	Coastal/inshore	(Parsons *et al.*, 2002))	PhD data	1990–1993	129		± 15	110-174	(Wilson *et al.*, 1999)
Bottlenose Dolphin	West coast of Scotland (including Barra)	Resident population	(Thompson *et al.*, 2011)	Scottish Government/SNH	2006–2007	45			33–66	(Cheney *et al.*, 2013)
Bottlenose Dolphin	Irish Sea (Cardigan Bay)	Coastal/inshore	(Parsons *et al.*, 2002)	SCANS II (uncorrected)	2005	397	0.23		362–414	(Evans, 2012b)
Bottlenose Dolphin	Connemara-Mayo, western Ireland	Coastal/inshore	(Mirimin *et al.*, 2011)	NPWS funded surveys	2014	c190				Simon Ingram, pers. comm.

Species	Assessment Units	Monitoring level	Reference	Baseline data collection, Survey	Baseline data collection Year	Abundance estimate	CV	SE	95% CI	Reference
Bottlenose Dolphin	Shannon Estuary	Resident population	(Mirimin *et al.*, 2011)	PhD data	1997	113		±16		(Ingram, 2000)
Bottlenose Dolphin	English Channel/Celtic Sea (Ireland, UK, France)	Coastal/inshore group	(ICES WGMME, 2013)	SCANS II	2005	4927	0.60		1662–14,608	(Hammond *et al.*, 2013)
Bottlenose Dolphin	Archipel de Molene (France)	Resident population	Ridoux *et al.*, 2000	Photo ID	1994–1998	45–47				(Le Berre and Liret, 2001)
Bottlenose Dolphin	Ile de Sein (France)	Resident population	Ridoux *et al.*, 2000	Photo ID	1992–1997	17				Liret (2001)
Bottlenose Dolphin	Northern Spain	Northern Spanish Cantabrian conti-nental waters, including offshore waters of the Bay of Biscay	(ICES WGMME, (2013)	Sightings from designed and non-designed surveys	2003–2011	10,687[5]	0.26		4,094–18,132	(López *et al.*, 2013)
Bottlenose Dolphin	Southern Galician Rias (NW Spain)	Resident population	(Fernandez *et al.*, 2011)	Photo ID	2000–10	> 255				(García *et al.*, 2011)
Bottlenose Dolphin	Coastal Portugal (out to 50 nm)	Coastal/inshore group	(ICES WGMME, 2013)	SafeSea and MarPro	2010	3051	0.78		294–31,666	(Santos *et al.*, 2012)
Bottlenose Dolphin	Sado Estuary	Resident population	(Gaspar, 2003, Fernandez *et al.*, 2011)	Census (from photo-id data)	1987	40				(dos Santos and Lacerda, 1987)
Bottlenose Dolphin	Strait of Gibraltar	Coastal/inshore group	(Giménez *et al.*, 2013)	Photo ID	2001–2008	297	0.06		276–332	(Chico Portillo, 2011)
Bottlenose Dolphin	Gulf of Cadiz	Coastal/inshore group	(Giménez *et al.*, 2013)	Photo ID	2009–2010	397	0.16		300–562	ICES WGMME (2013) (MAGRAMA 2012)

[5] Uncorrected for animals missed on the transect line.

Species	Assessment Units	Monitoring level	Reference	Baseline data collection, Survey	Baseline data collection Year	Abundance estimate	CV	SE	95% CI	Reference
Bottlenose Dolphin	Oceanic Waters	Oceanic group	(Louis *et al.*, 2014)	SCANS II & CODA	2005/2007	11,923	0.21		7,935-17,915	Macleod *et al.*, 2009; Hammond *et al.*, 2013)
Bottlenose Dolphin	The Azores – central group	Coastal/inshore	(Silva *et al.*, 2008)	Photo-id (mark re-capture survey)	2003	Adults: 312 Subadults: 300	A:0.06 S:0.10		A:254–384 S:232–387	(Silva *et al.*, 2009)
White-Beaked Dolphin	Britain and Ireland	AU	(Banguera-Hinestroza *et al.*, 2009)	SCANS II	2005	15,895	0.29		9,107–27,743	(Hammond *et al.*, 2013)
Minke Whale	NE Atlantic stock (OSPAR region II and III)	AU		SCANS II/CODA	2005/2007	23,163	0.27		13,772 – 38,958	(Macleod *et al.*, 2009, Hammond *et al.*, 2013)
Minke Whale	Continental Portugal until the 50 nm (OSPAR region IV)	Coastal/inshore		MarPro	2011	2919 (0,039 ind/km^2)	0.434		1247–6834 (0,0167–0,0913)	(Santos *et al.*, 2012)
Common Dolphin	NE Atlantic	Population	(Murphy *et al.* 2009)	SCANS II, CODA,	2005/2007	174,485	0.26		105,694-288,048	(Macleod *et al.* 2009, Hammond *et al.* 2013)
Striped Dolphin	NE Atlantic	Population	(ICES WGMME 2014)	CODA	2007	61,364	0.93		12,323-305,568	Macleod *et al.* 2009

As noted in Section 10.2, Meesters *et al.* (2007) undertook power analysis to identify the effectiveness of the existing survey schemes relative to a specific conservation target, i.e. Wadden Sea harbour seal EcoQO subunit (Meesters *et al*., 2007). Results highlighted that the current monitoring programme was insufficient to meet the OSPAR EcoQO guidelines (80% power and 5% probability to detect a change in abundance of minimally 10% over ten years). The Wadden Sea harbour seal monitoring programme had sufficient power (80%) to detect a minimal trend of 2.2 % per annum in ten years and 6% per annum in six years. It was noted that as the targets were not even met with annual surveying, undertaking biennial or triennial surveys cannot be justified. To meet the set requirements it was suggested that monitoring should be increased to at least four simultaneous counts in August throughout the entire Wadden Sea (Meesters *et al*., 2007). Prior to this, recommendations were for at least three surveys carried out annual during the late pupping season to enable assessment of pup production in that specific year, and two extra surveys every five year to detect a possible shift in timing of the pupping season and hence the moult (Meesters *et al*., 2007). The current monitoring programme however only undertakes two replicate counts in August.

During this year's meeting, simple power analyses were conducted using software TRENDS (Gerrodette, 1993) to investigate the rate of decline in grey seal relative abundance that could be detected from SMRU biennial grey seal pup surveys in the UK. SMRU pup surveys generate estimates of total pup production with a CV of about 0.1 and now occur every two years. The probability of making a Type-I error was set at alpha = 0.05. The probability of making a Type-II error was set at beta = 0.20; equivalent to a power of 80%. Table 10.6 shows the minimum detectable rate of decline per year for biennial surveys over periods of six and twelve years (one and two MSFD reporting periods). A lower annual rate of decline of 3% was detectable over a longer time period (12 years) with a larger number of surveys.

Table 10.6. Minimum detectable rate of decline per year for biennial surveys over periods of six and twelve years.

Survey interval (yrs)	Monitoring period (yrs)	Number of surveys	CV of pup production	Minimum annual rate of decline detectable
2	6	4	0.1	10%
2	12	7	0.1	3%

The WG has proposed two grey seal assessment units within OSPAR regions II, III and IV: (1) North Sea and (2) western Britain, Ireland and western France. As this will involve integration of data from three or more national monitoring schemes, data collection techniques should be assessed to determine whether it is possible to sum totals for pup production estimates in order to derive overall trends for both management assessment areas, and as such provide for an assessment for both indicator M-3 and M-5.

During the WGMME meeting, the ICES seal database was reviewed taking into consideration requirements of the OSPAR MSFD indicator assessments. In its current format, the database is not sufficient for assessment of MSFD indicators measuring temporal trends of abundance and pup production. Primarily as only limited information on grey seal pup counts, and harbour seal moult counts have been collated, with no corresponding metadata information, SEs or CVs, and whether the data represent exact numbers or estimates. Data from replicate surveys would also be desirable for increasing the power of abundance trend analysis based on moult counts (see ToR E). Required data and metadata to undertake this task have been identified, resulting in a proposal for a new reporting format. ICES proposes that Member States, and for completeness also Norway, be requested to populate this seal database for assessment purposes by the end of 2015 (OSPAR testing phase), by the end of 2016 (OSPAR intermediate assessment of indicators in 2017) and for the coherent assessment required under the MSFD during the years there after.

The Special Committee on Seals (SCOS; UK) will provide an initial assessment of data availability and suitability in the UK to meet the seal indicators and targets described in the OSPAR indicator summaries. This work will be undertaken during 2014 with publication expected in 2015. If this document is available, results will be reviewed by the WGMME in 2015. It will be investigated if the assessment can be extended to the relevant assessment units as agreed by OSPAR.

The WG **recommends** that for each AU, baselines be ascertained and an assessment of growth rates / longitudinal abundance data is undertaken prior to the establishment of targets. This assessment will inform on the relative status of each AU, i.e. if individuals are close to carrying capacity.

10.3.6 Seals: baselines: M-3, M-5

The adoption of baselines of seal abundance for evaluation of indicator targets was discussed extensively by the WG. It was decided that historic or natural abundances of seal populations are generally unknown and that it is supposed that most populations are currently far from such levels of abundance. With this lack of a meaningful baseline, the WGMME recommends a number of approaches, depending on the species/assessment units:

a) For grey seal assessment units in the North Sea, 1984 is proposed as the baseline, i.e. the year that current UK monitoring activities were instigated. At this time, the large majority of grey seals in the North Sea were breeding along the UK coastline, and abundance levels in other North Sea countries were at their lowest, relative to current estimates. Grey seals disappeared from the eastern North Sea around 1500 AD, and breeding did not occur again until the end of the 1970s when a colony was established in the German Wadden Sea. Following 1984, there has been a consistent increase in UK grey seal pup production estimates in the North Sea (see Figure 10.12).

b) As the rate of increase of grey seals off the Scottish west coast (inner and outer Hebrides) has stabilised (SCOS, 2012; Table 10.7), this suggests that animals inhabiting this region are at carrying capacity and to use a current baseline (see Figure 10.13).

c) For harbour seals, baselines should be set for each specific assessment unit, after assessing longitudinal abundance data (see Figures 10.13, 10.14).

d) Adoption of 'rolling' baselines could be undertaken, where the abundance of seals in assessment units is assessed relative to data from the previous e.g. five or ten-years, depending the power of the monitoring.

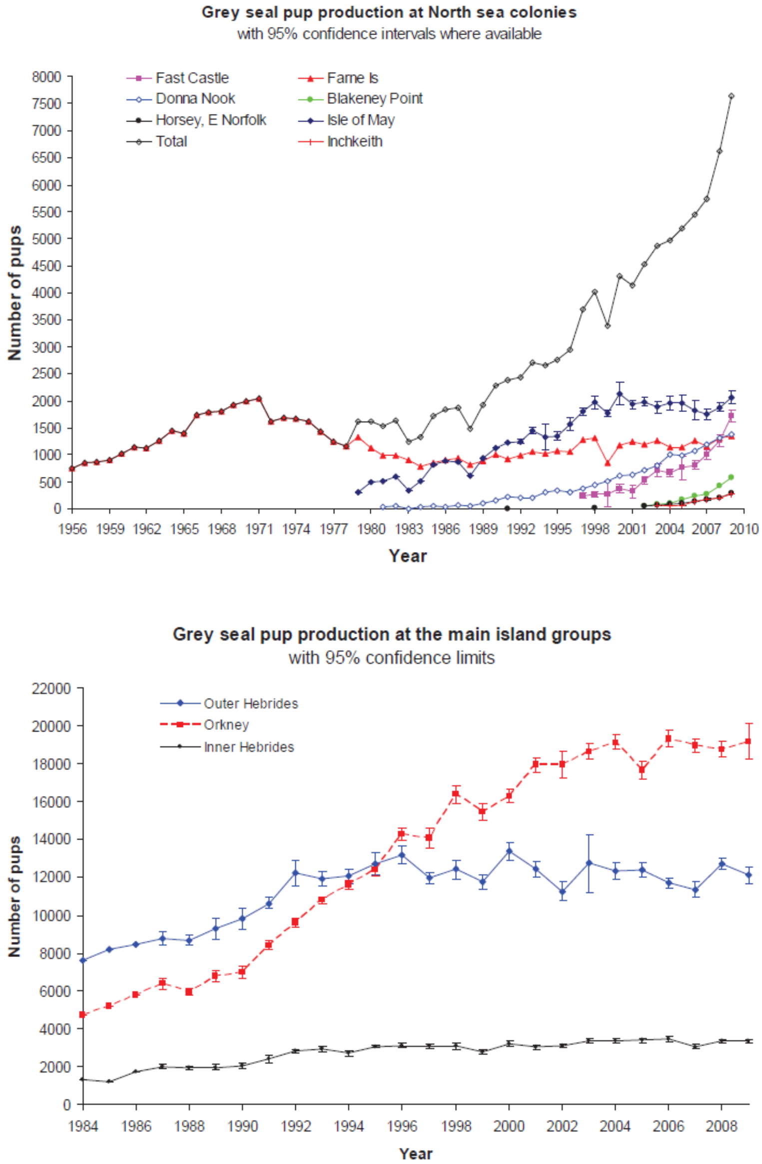


Figure 10.12. Trends in UK grey seal pup production at North Sea colonies. Production values are shown with their 95% confidence limits where these are available. These limits assuem that the various pup development parameters involved in the estimation procedure remain constant from year to year. Although they therefore underestimate variability in the estimates, they are useful for comparing the precision of the estiamates in different years (Taken from (SCOS, 2011). The long-term average rates of change suggest that the growth of pup production in the Inner and Outer Hebrides has effectively stopped with little change in the Inner Hebrides and possibly a small decrease in the Outer Hebrides since the mid-1990s. The rate of increase in pup production in Orkney also appears to have reduced since the end of the 1990s (SCOS, 2012).

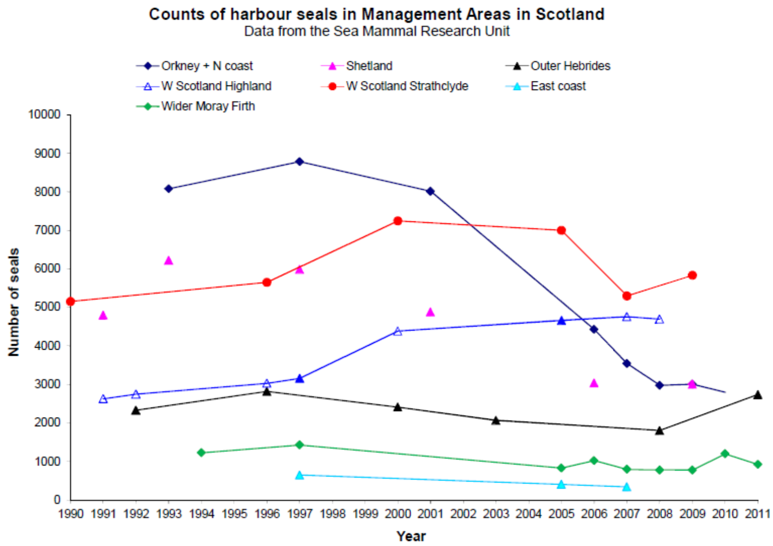


Figure 10.13. Trends in moult counts of harbour seals around Scotland (Taken from SCOS, 2012).

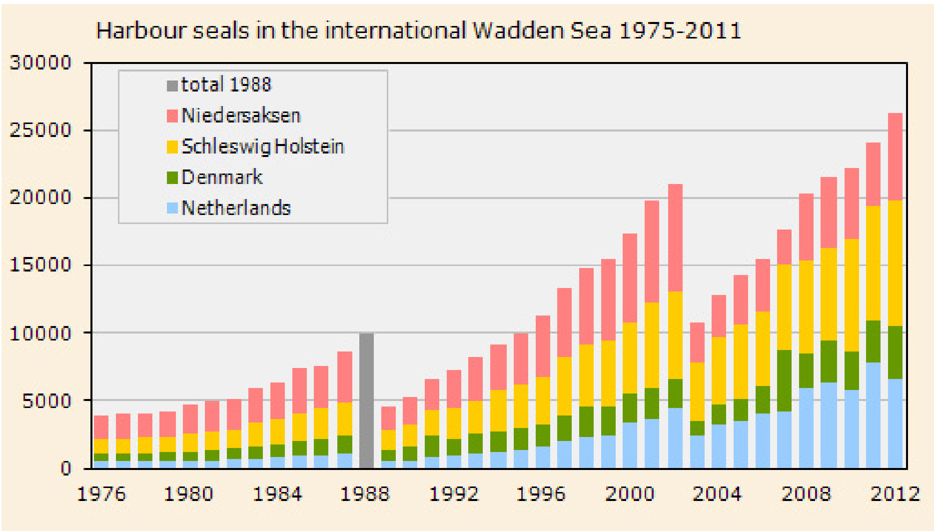


Figure 10.14. Numbers of harbour seals in the Wadden Sea assessment unit[6].

Approximately 30% of European harbour seals are found in the UK; this proportion has declined from approximately 40% in 2002 (SCOS, 2012). Counts in the Wash and eastern England did not demonstrate any recovery from the 2002 epidemic until 2009 but have increased greatly in recent years (SCOS, 2012). Major declines have now been documented in several harbour seal populations around Scotland, with declines since 2000 of 68% in Orkney, 50% in Shetland, and 90% in the Firth of Tay. However the pattern of declines is not universal and the recorded declines are not thought to have been linked to the 2002 PDV epidemic that seems to have had little effect on harbour seals in Scotland (SCOS, 2012). One factor possibly attributing to the cause of

[6] Taken from http://www.zeeinzicht.nl/vleet/index.php?id=4171&template=template-vleeteng&language=2&item=Harbour-seal.

the decline has been competition for resources with grey seals. If this is the case, then the independent targets for both seal species will need to be re-assessed.

The average growth rate for the Wadden Sea harbour seal population for the first five years after the 2002 epidemic was 13.4%, just above the theoretical maximum annual growth rate of 13% per year for this species (Härkönen *et al.*, 2002; Trilateral Seal Expert Group (TSEG), 2013). Since then, the sliding 5-year average growth rate has been decreasing (11.1% in 2009, 9.3% in 2010, 9.0% in 2011, 8.3% in 2012 and 5.8% in 2013). This decreasing growth rate may be due to the population approaching carrying capacity, or because the moulting peak has shifted over the years. This needs to be investigated by exploring pup survival and the timing of the peak of the moult (Trilateral Seal Expert Group (TSEG), 2013).

The UK grey seal population represents approximately 38% of the world population on the basis of pup production. The other populations in the Baltic and the western Atlantic (though not the Gulf of St Lawrence) are increasing at a faster rate than the UK (see Table 10.8; SCOS, 2012). Bowen *et al*. (2003) reported that grey seals on Stable Island in Nova Scotia, Canada increased exponentially at an annual rate of 12.8% for four decades in the face of considerable environmental variability.

Table 10.7. Grey seal pup production estimates for the main colonies surveyed in 2010 and available 2012 grey seal pup production estimates compared to UK wide estimates (Taken from SCOS, 2013).

Location	Pup production in 2012	Average annual change from 2010 to 2012	Pup production in 2010	Average annual change in pup production from 2005 to 2010
Inner Hebrides	4,027	+9.0%	3,391	-0.0%
Outer Hebrides			12,857	+1.0%
Orkney			20,312	+2.4%
Firth of Forth	5,175	+10.0%	4,279	+9.0%
All other Scottish colonies (incl. Shetland & mainland)			3,299[1]	
Total Scotland			**44,138**	**+1.9%**
Donna Nook + East Anglia	3,359	+14.4%	2,566	+15.0%
Farne Islands	1,603	+3.4%	1,499	+5.9%
SW England (last surveyed 1994)			250	
Total England			**4,315**	
Total Wales [2]			**1,650**	
Total England & Wales			**5,965**	**+6.7%**
Northern Ireland			100	
Total (UK)			**50,203**	**+2.4%**

[1]Estimate derived from data collected in different years
[2]Estimate from indicator sites in 2004-05, multiplier derived from 1994 synoptic surveys

Table 10.8. Relative sizes of grey seal populations. Pup production estimates are used because of the uncertainty in overall population estimates (Taken from (SCOS, 2013).

Region	Pup Production	Year	Possible population trend[2]
UK	**50,200**	2010	Increasing
Ireland	1,600	2005	Unknown[1]
Wadden Sea	430	2012[3]	Increasing [2]
Norway	1,300	2008	Increasing[4]
Russia	800	1994	Unknown[2]
Iceland	1,200	2002	Declining[2]
Baltic	4,700	2007	Increasing[2,5]
Europe excluding UK	**10,030**		Increasing
Canada - Sable Island	62,000	2010	Increasing[6]
Canada - Gulf St Lawrence + Eastern Shore Canada	14,200	2010	Declining[7]
USA	2,600	2008	Increasing[8]
WORLD TOTAL	**129,000**		Increasing

[1] Ó Cadhla, O., Strong, D., O'Keeffe, C., Coleman, M., Cronin, M., Duck, C., Murray, T., Dower, P., Nairn, R., Murphy, P., Smiddy, P., Saich, C., Lyons, D. & Hiby, A.R. 2007. An assessment of the breeding population of grey seals in the Republic of Ireland, 2005. Irish Wildlife Manuals No. 34. National Parks & Wildlife Service, Department of the Environment, Heritage and Local Government, Dublin, Ireland.
[2]Data summarised in:- Grey seals of the North Atlantic and the Baltic. 2007. Eds: T. Haug, M. Hammill & D. Olafsdottir. NAMMCO Scientific Publications, Vol. 6.
[3] Brasseur, S., Borchardt, T., Czeck, R., Jensen, L.F., Galatius, A., Ramdohr, S., Siebert, U., Teilmann, J., 2012, Aerial surveys of Grey Seals in the Wadden Sea in the season of 2011-2012 - Increase in Wadden Sea grey seals continued in 2012. Trilateral Seal Expert Group.
[4] Øigård, T.A., Frie, A.K., Nilssen, K.T., Hammill, M.O., 2012, Modelling the abundance of grey seals (*Halichoerus grypus*) along the Norwegian coast. ICES Journal of Marine Science: Journal du Conseil, 69(8) 1436-1447.
[5] Baltic pup production estimate based on mark recapture estimate of total population size and an assumed multiplier of 4.7 HELCOM fact sheets (www.HELCOM.fi)
[6] Bowen, W.D., McMillan,J.I. & Blanchard, W. 2007. Reduced Population Growth Of Gray Seals At Sable Island: Evidence From Pup Production And Age Of Primiparity. Marine Mammal Science, 23(1): 48–64
[7] Thomas,L.,Hammill,M.O. & Bowen,W.D. 2011 Estimated size of the Northwest Atlantic grey seal population 1977-2010 Canadian Science Advisory Secretariat: Research Document 2011/17 pp27.
[8]NOAA (2009) http://www.nefsc.noaa.gov/publications/tm/tm219/184_GRSE.pdf

10.4 Current monitoring practices for seals and cetaceans in OSPAR Regions II, III, and IV

Table 10.9 summarises current national monitoring programmes for seals and cetaceans.

Table 10.9. Current national monitoring schemes for cetaceans. (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	Sightings[7]	Strandings	Fishery bycatch/directed catches[8]	Biological data[9]	Other[10]
Belgium	P/I: Monitoring density and distribution: aerial surveys; additional information from seabird surveys; P/I/G (offshore windfarms); G/ I/V: reporting of opportunistic sightings by the public through dedicated Internet sites set up by scientific institute and NGOs. Only decidated harbour porpoise monitoring - 4 aerial surveys/year since April 2008. Mainly between February and November.	G/ 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. +50 harbour porpoises PME/year, other species *ad hoc*, but very few	G/I/V: 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. No dedicated on board bycatch monitoring programme; ad hoc indirect assessment of bycatches through the investigation of stranded animals	G/I: research only on bycaught and stranded animals, no biopsies taken.	P/I/G: Assessment of distribution and density is carried out mainly in the framework of offshore windfarm projects; dedicated monitoring through aerial surveys, passive acoustic monitoring (C-PoD). - 3–4 PoDs moored 10 months per year
Denmark	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	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: No bycatch estimate is available from the North Sea (see Vinther and Larsen 2004[11]). 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	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

[7] Monitoring of abundance, distribution, movements, behaviour.

[8] Where relevant.

[9] From carcasses of stranded or bycaught animals or from biopsies (diet, life history, pathology, contaminants).

[10] For example, tagging and photo-ID studies, monitoring related to ship strikes, disturbance, naval exercises, seismic surveys, wind farms, etc.

[11] 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	SIGHTINGS[7]	STRANDINGS	FISHERY BYCATCH/DIRECTED CATCHES[8]	BIOLOGICAL DATA[9]	OTHER[10]
	survey. Current harbour porpoise monitoring 2011–2015: Skagerrak and the southern Danish North Sea are covered by an annual aerial survey in July. In 2012, a visual ship-based survey covering the entire inner Danish waters management unit was performed, by methods to yield results comparable to the previous SCANS surveys (1994 and 2005). 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			not studied.	1997. G = In the inner Danish waters, the six largest habitat areas for porpoises are monitored by stationary acoustic porpoise detectors (C-PODs). In 2011 and 2013, ship-based acoustic surveys of 11 habitat areas. 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.
France	G/I: Annual surveys on oceanographic vessel during spring and automn in Bay of Biscay, during winter in English Channel P/I: Two large-scale aerial survey in Bay of Biscay and English Channel in 2012 (one in summer, one in winter). P: Monthly survey on ferry boat in North sea between Calais and Douvres I/V: Reporting of opportunistic sightings by the public.	G/ I/V: National Stranding Network coordinated since 1972 / Annual report in French. Marine mammal tissue bank since 2000.	G/I: On-board bycatch monitoring programm G/I: Indirect assessment of bycatches through the investigation of stranded animals V: Reporting by fishermen in few cases	P/I: Sampling from stranded animals for determination of age, maturity and diet. Samples also taken for other analyses (i.e. isotopes, contaminants, genetics, parasites, etc.). P: Biopsies collected on bottlenose dolphins Sample analysis is	P: Photo-ID programs on Bottlenose dolphins (and long-finned pilot whales). P: Deployement of CPOD since 2012 to monitor harbour porpoise

Country	Sightings[7]	Strandings	Fishery bycatch/directed catches[8]	Biological data[9]	Other[10]
				carried out through collaborations, as part of PhD, MSc students projects and specific projects.	
Germany	G/P = several projects G = Two large-scale monitoring schemes in the North Sea; (a) Dedicated aerial surveys for assessing the distribution and density of harbour porpoise in the German part of the North Sea since 2002 in the framework of the construction of windmill parks, and to investigate potential areas for implementing Natura 2000 (Scheidat *et al.*, 2004; Scheidat *et al.*, 2007; Gilles and Siebert, 2009; 2010; Gilles *et al.*, 2009; 2011; 2012). (b) 2008–2012: dedicated sightings surveys in the southwestern part of the German North Sea and parts of neighbouring Dutch waters (10 934 km^2) in the framework of the "StUKplus-Project" dealing with the monitoring of the offshore wind test field "Alpha Ventus". These include both aerial surveys (3–5 times per year) in the wider testfield areas in 2008–2012; shipboard surveys in a smaller area (2110 km^2) around the windfarm site (DP BT SCANSII method and acoustic) in 2008–2011; and C-POD stations from 2008 until February 2012. The overall project will continue until the end of 2013 (Siebert *et al.*, 2011; Dähne *et al.*, 2012; Gilles, pers. comm.)	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 = 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.	
Ireland	IWDG Sighting scheme includes casual and effort related sightings data with 22 000 records available online (www.iwdg.ie) with 17 500 records since	Irish Whale and Dolphin Strandings Scheme, running since 1991. Annual report published in Irish	IWDG completed independent observer study of the Celtic Sea Herring Fishery in 2012	Post-mortem examination is carried out on a sample of stranded cetaceans	Three year study 2009–2011) to develop acoustic monitoring techniques for reporting obligations

Country	Sightings[7]	Strandings	Fishery bycatch/directed catches[8]	Biological data[9]	Other[10]
	2003 (Berrow *et al.*, 2002; 2010) (V, G) Dedicated line rtransects suirveys of eight sites for harbour porpoise carried out since 2008 using distance sampling to estimate densities and abundance (Berrow *et al.*, 2007; 2013; 2014). Over 37 days at sea, a total of 475 track-lines were surveyed for a total distance of 20 623 km. From the 332 sightings, a total of 618 individual harbour porpoise were recorded. Overall density estimates ranged from 0.53 to 2.03 porpoises km^2 (without correction for g(0)). Mean group size varied from 1.41 to 2.67 (G). Dedicated nearshore surveys (between 6–12 nmls offshore) carried out at six sites between in 2010 and 2012 (Ryan *et al.*, 2010; Berrow *et al.*, 2011; Berrow *et al.*, 2012) to estimate sighting rates and density. A total of 1013 km of track-line were surveyed in sea-state ≤3, with 189 sightings, comprising 793 individuals of six species. Harbour porpoise were the most widespread species occurring at five of the six sites. Minke whale occurred at four sites, common dolphin at three sites and Risso's dolphin at two sites (G). Offshore surveys since 2004 under variety of projects including ISCOPE and PReCAST, recently published in offshore atlas (Wall *et al.*, 2013), which included data of 3300 sightings of 35 000 individuals (16 species) collected during 1089 days at sea (V, P, G). Five dedicated cetacean surveyshave been carried out since 2008 targeting shelf edge habitats and using single and double platform methodology and distance	Naturalists Journal (O'Connell and Berrow, 2013; in press). Data available on line at www.iwdg.ie (V, G). Record number of strandings in 2013 with a total of 193 strandings of 204 individual animals were received, an increase of almost 20% on 2012, which itself was a 4% increase on 2011.	funded by the Celtic Sea Herring Management Advisory Committee (P). A total of 20 fishing trips carried observers and a total of 35 hauls were ob-served. A total of 858 tonnes of herring were landed which accounted for 5.1% of the TAC for the season. No marine mammal bycatch was recorded (McKeogh and Berrow, 2014). During 2011–2012 pelagic fishing season, 15 trips carried observers who witnessed 69 fishing events or hauls (Boyd *et al.*, 2012), in compliance with Bycatch Regulation 812/2004 (G). Similar project carried out in 2011–2012 (McCarthy *et al.*, 2011) (G).	each year, depending on specific research projects but their is no official PM project (V, I). Stranded cetaceans sampled for skin for storge in the Irish Cetacean Genetic Tissue Bank hosted the the National Museum of Ireland (Natural History) and ran as a collaboration between NMI(NH) and IWDG (V). Samples from around 400 inividuals are stored to date and have been accessed by a number of projects including studies on bottlenose dolphins, bottlenosed whales, striped dolphins and long-finned pilot whales	(O'Brien *et al.*, 2013b) (G). Static Acoustic monitoring of bottlenose dolphins in the Lower River Shannon cSAC since 2001. Four sites currently monitored with time-series as far back as 2007 (V, I). Acoustic monitoring of delphinids in Broadhaven Bay, Mayo since 2007 (P)

Country	Sightings[7]	Strandings	Fishery bycatch/directed catches[8]	Biological data[9]	Other[10]
	sampling as well as PAM (Wall *et al.*, 2009; 2010; Ryan *et al.*, 2011; O'Brien *et al.*, 2013a; 2014)(V, P) IWDG carry out monthly ferry surveys on three routes from Ireland (incl. Northern Ireland) to the UK since 2001 (V). Marine mammal monitoring Broadhaven Bay, Mayo since 2002 (P).				
The Netherlands	Proeject-based aerial surveys for estimating harbour propoise abundance started in 2008. Different projects: land-based observations, ESAS observations IMARES: aerial 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).	Collected by different organizations (V, G); animals are brought together at the University of Utrecht for further analyses.	IMARES (G): short-term monitoring and pilot projects e.g. using camera systems to monitor bycatch (Bram Couperus).	University of Utrecht (G).	
Portugal	P&I - Dedicated aerial surveys using distance sampling (survey area ranges from the coast to the 50 nm) 2010 SafeSea project - Two pilot campaings in summer/autumn 2011 until 2015 Life MarPro, one campaign per year in autumn. P&I - Dedicated offshore vessel survey. 2011 Life MarPro using distance sampling. Only one survey in Summer 2011 covering the continental Portuguese area between 50 nm and the 220 nm	G, P & I - National Strading Network coordinated by ICNF with 3 regional networks: North (coordinated by SPVS), Center /South (ICNF) and Algarve (SPVS). New dedicated necropsy facilities in Algarve and Quiaios. MATB - Marine Animal Tissue Bank since 2000 coordinated by SPVS, with samples of more than 1200	P&I – On board bycatch monitoring programme in purse-seine, polyvalent, longline and trawler fleets. Voluntary declaration scheme with logbooks and Electronic monitoring using video cameras on-board selected vessels. 2010 SafeSea project 2011 until 2015 Life MarPro	P&I = stomach content: analysis in the most common stranded species. Fatty acid and stable isotopes mainly in odontocetes Age determ. in the most common stranded species Pathology: all live stranded cetaceans and in very fresh dead stranded animals	I - accoustic monitoring of harbour porpoise near set-nets using different accoustic methods

COUNTRY	SIGHTINGS[7]	STRANDINGS	FISHERY BYCATCH/DIRECTED CATCHES[8]	BIOLOGICAL DATA[9]	OTHER[10]
	P&I - Vantage point Surveys with monthly effort (mainly dedicated to Harbour Porpoise) -2008–2010 SafeSea project, 16 vantage points -2011 to 2015 Life MarPro,.24 vantage points P&I - Surveys in platforms of opportunity using strip-transect approach. 2008–2010 SafeSea project 2011 until 2015 Life MarPro G&P regular surveys (since 2011) in the maritime area adjacent to the Sado estuary to monitor both resident and coastal populations of bottlenose dolphins	individuals. An average necropsy number of about 150 animals per year in the last five years		Contaminants: in odontocete species. Relation between life-history data, pathologies and contaminant burdens -CetSenti Project FCT (Compete-FEDER)	
Azores (Portugal)	P: Dedicated surveys under different projects, 1999–present, mostly spring-summer, Centre of IMAR (Marine Research Institute) of the University of the Azores (IMAR-DOP/UAç). Spatial coverage varied between surveys, G&I: Azores Fisheries Observer Programm (POPA), 1998-present, April-October, IMAR-DOP/UAç. Covers all archipelago and offshore seamounts. P: Passive acoustic monitoring, 2007-present, year-round, IMAR-DOP/UAç. Whale-watching operators, 1993–present, April–October, private funding. I: Small-scale surveys, 2004-present, Nova Atlantis Foundation.	G&I: Azorean Cetacean Stranding Network, 1996–present . Detailed necropsies are not conducted due to absence of veterninarians and lack of adequate conditions. Routine collection of biological samples is undertaken in a few islands. Minke whale: total seven strandings/total two necropsies (1996–2010) Common dolphin: average four strandings year/average three	G&I: Azores Fisheries Observer Programm (POPA), 1998-present, IMAR-DOP/UAç. POPA covers ~50% of the tuna-fishery fleet. Coverage of other fisheries is lower (see Silva *et al*., 2011). Bycatch estimate is available for the tuna-fishery (Silva *et al*., 2002, 2011). P: Short-term & irregular monitoring programmes of several fisheries (mainly surface and bottom longline), 1990–present,	P: Diet/foraging ecology: stomach contents, fatty acids and stable isotopes (from biopsies and strandings), 2002–present, IMAR-DOP/UAç. P: Genetic structure (from biopsies and strandings), 2002–present, IMAR-DOP/UAç. Pathology: not collected routinely	P: Photo-id, 1999–present, IMAR-DOP/UAç. Photo-id, whale-watching operators 2000–present, private funding. V: Photo-id, Nova Atlantis Foundation, 2004–present. P: Disturbance from Whale-watching, 1998–2006I, MAR-DOP/UAç. I: Disturbance from Whale-watching, 2004, Nova Atlantis Foundation. P: Tagging, 2008–present,

Country	Sightings[7]	Strandings	Fishery bycatch/directed catches[8]	Biological data[9]	Other[10]
	P: dedicated survey to bottlenose dolphins, 2002–2006, mostly spring-summer, IMAR-DOP/UAç.	necropsies year (1996–2010) Bottlenose dolphin: average 0.8 strandings year/average 0.8 necropsies year (1996–2010).	DOP/UAç. Low observer coverage (see Silva *et al.*, 2011.		IMAR-DOP/UAç. P: Acoustic, 2007–present, IMAR-DOP/UAç.
Spain	Patchy coverage. Dedicated anual aerial surveys using distance sampling in Valencia (Mediterranean) and Andalucia (southern Spain) (P&G). In Galicia, pilot aerial surveys were carried out in a few years covering the southern half of the coast. No large-scale dedicated surveys since SCANS II, CODA (P). Small-scale dedicated surveys (V&P) (in Galicia and northern Spain and Gulf of Cadiz, also off Murcia, Andalucia, the Balearic Islands and the Canary islands). Long-term medium scale monitoring of abundance and distribution (northern Alborán Sea) (P) but since 2011 sporadically. P&V – Coastal observations: from 30 vantage points in Galicia visited once per month (ongoing since 2003). P&I - Surveys in oceanographic vessels using distance sampling since 2007 in north and northwestern waters in spring. Data collected opportunist-ically year-round from whale watching vessels in Canary Islands and southern Spain.	Not nationally coordinated, different stranding networks operating along the coast: e.g. Galicia: since 1990 (V & G since 1999), in Asturias (northern Spain, V&P, since 1998), in the Basque Country (V&P since 1999), in Andalucia (southern Spain) since 2004 (G) , since 1990 (G) in Valencia. Two separate NGOs run strandings monitoring schemes in the Canary Islands (G&P).	No dedicated on board bycatch monitoring programme (a pilot programme ran for 1 year); observers of the discards monitoring programe collectinformation on protected species bycatch (through AZTI and IEO); ad hoc indirect assessment of bycatches through the investigation of stranded animals. In Galicia, two interviews surveys carried out (e.g. Goetz *et al.*, in Press) and voluntary reporting (V/P)	Necropsies are carried out only on fresh or relatively fresh specimens. Samples taken for determination of age, maturity and diet (mainly stomach conten ts) (P&I). Samples also taken for other analyses (i.e. isotopes, contaminants, genetics, parasites, etc.) (P&G&V). Sample analysis is carried out through collaborations, as part of PhD, MSc students projects and specific projects.	Photo-ID programes for coastal, resident Tursiops population in Galicia (G&P&V), Gulf of Cadiz (P) and the strait (P). Also for Risso´s dolphins, common dolphins and long-finned pilot whales in Galicia (P&V). For bottlenose dolphins in the Basque country (P&V). For common dolphin, orcas, long finned pilot whales, sperm whales in Souther Spain (Gulf of Cadiz, Gibraltar and Alboran Sea) (P&V). For sperm whales and other species in the Balearic sea (P&V). For boltlenose dolphins and other species in the Canary Islands (P&V). ARGOS tagging on orcas and long finned pilot whales in the Gulf of Cadiz Passive acoustic monitoring: in Galicia isolated experiences

Country	Sightings[7]	Strandings	Fishery bycatch/directed catches[8]	Biological data[9]	Other[10]
					(using T-PODs) (P&V), more regular in coastal waters of the Balearic islands since 2009 (using PODs and EAR) (P) Some acoustic monitoring also takes place during sighting surveys
Sweden	Irregular P. No monitoring effort has been conducted in Swedish North Sea waters since SCANS II in 2005.	Continuous reporting G	Irregular P.	Pathology G, P	
UK	P, G: Large-scale distance sampling population surveys and passive acoustic surveys: SCANS II (Hammond *et al*., 2013) and CODA (2009); G: Large-scale population surveys: ESAS. Data are collected from four of the twelve IBTS cruises per year which cover the North Sea (two in winter and two in summer). G: Regional Sea dedicated aerial surveys (southern North Sea) in 2011 and 2013 primarily for harbour porpoise but also recording other species (e.g. Gillis *et al*., 2012; Geelhoed *et al*., 2014). G,P: Local scale line transect surveys in Welsh waters (e.g. Veneruso, and Evans, 2012) V: Ferry surveys undertaken by Atlantic Research Coalition (ARC) established in 2001 and comprises eight organisations which regularly conduct fixed-route transect surveys on 20 commercial ferry routes throughout northwest European waters using effort-related and standardised scientific	I, G - 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; Deaville and Jepson, 2012). V- local collation of records by some voluntary schemes (e.g. Cornwall Wildlide Trust, see http://www.cwtstrandings.org) Approx 100 PME undertaken per year, with the majority being harbour	G, P: In 1992 a project to assess marine mammal bycatch in gillnet fisheries in the Celtic Sea was established. 1994–1997, independent monitoring scheme on UK gill and tanglenet vessels throughout the North Sea and west of Scotland was introduced. As this work continues and evolved, the UK Bycatch Monitoring scheme became formally established in 2005 and continues today. The monitoring effort allows for a precise estimate of the total bycatch, which can be identified to gear and fishery (e.g. Northridge *et al*., 2011; 2012; 2013).	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 (e.g. Law *et al*., 2014). A national cetacean	G, P - Moray Firth SAC site condition monitoring using photo ID surveys. G,P,V; Wider Cardigan Bay bottlenose dolphin Photo ID surveys (e.g. Veneruso, and Evans, 2012) V: Risso's dolphin photo ID work (e.g. de Boer *et al*., 2013) G, P - Joint Cetacean Protocol (JCP) project (see http://jncc.defra.gov.uk/page-5657) was initiated in 2006. The JCP assembled disparate effort-related cetacean sightings datasets from northwest European Atlantic waters and included those from all major UK sources e.g. SCANS I & II; CODA

COUNTRY	SIGHTINGS[7]	STRANDINGS	FISHERY BYCATCH/DIRECTED CATCHES[8]	BIOLOGICAL DATA[9]	OTHER[10]
	recording methods. G, V: systematic land-based visual surveys. In Northern Ireland sites were set up at equidistant points along the coast and not based on previous observations of animal presence. Data collected on a monthly basis throughout the year. Data from voluntary organisations collated by Sea Watch Foundation.	porpoise and common dolphin.		tissue archive and a web-accessed strandings/pathology database is maintained by CSIP partner organizations. P- additional funding for some aspects of above.	surveys; ESAS; SWF; Atlantic Research Coalition (ARC); and from other non-governmental and marine renewable industry sources. Further development ios required, but it is envisaged that the project could be used to identify trends in distribution and relative abundance at scales relavant to HD and MSFD reporting requirements. (e.g. Paxton and Thomas, 2010; Paxton *et al*., 2011; Paxton *et al*., 2013). Similar work has been undertaken for Welsh waters (Baines and Evans, 2012) P; Changes in distributional pattern in relation to noise: short-term impacts through use of static passive acoustic devices (PODs) and visual data collected from digital aerial surveys and land-based surveys (e.g. Thompson *et al*., 2013). G, I: Marine Mammal Obersever records collected as part of the consents process for seismic surveys are collated by JNCC .

Country	Sightings[7]	Strandings	Fishery bycatch/directed catches[8]	Biological data[9]	Other[10]
					G, P: the cumulative impact of cetacean exposure to noise is being developed through the Population Consequences of Disturbance (PCoD) project. DEPONS (*Disturbance Effects on the Harbour Porpoise Population in the North Sea*) project aims to validate the individual based simulation model for analysing the population consequences of anthropogenic disturbances.

Table 10.10. Current and known plans for monitoring harbour seals.

Country	MSFD Assessment units	Current monitoring	Monitoring method	Comments
United Kingdom	Shetland	Population monitoring: moult	Single aerial survey on approximate 5 yearly schedule	Minimum required
	Orkney and North Coast	Population monitoring: moult	Single aerial survey on approximate 5 yearly schedule	Minimum required
	Moray Firth	Population monitoring: breeding and moult	Repeat aerial survey, annual	
	East coast Scotland (Firth of Tay only) [12]i	Population monitoring: moult	Single aerial survey, annual	Minimum required
	Northeast England	No formal monitoring as population is very low		
	Southeast England	Population monitoring: breeding and moult	Repeat aerial survey, annual	
	West England and Wales	No formal monitoring as population is very, very low		
	Southwest Scotland	Population monitoring: moult	Single aerial survey on approximate 5 yearly schedule	Minimum required
	West Scotland	Population monitoring: moult	Single aerial survey on approximate 5 yearly schedule	Minimum required
	Western Isles	Population monitoring: moult	Single aerial survey on approximate 5 yearly schedule	Minimum required
	North Ireland	Population monitoring: moult	Single aerial survey on approximate 5 yearly schedule	Minimum required
Netherlands	Delta	Extension of bird surveys	Aerial survey, monthly	No formal assessment yet. Numbers of animals in this area is thought to be low.
Netherlands/ Germany/ Denmark	Wadden Sea	Population monitoring: breeding and moult	Repeat aerial survey Annual	

[12] Remainder of east coast as per all other areas; every five years approximately.

Germany	Helgoland	Population monitoring	Daily land counts
Denmark	Limfjord	Population monitoring: moult	Repeat aerial survey Annual
Norway/Sweden	Northern Skagerrak and Oslo Fjord	Population monitoring: Moult	Aerial survey, every annual
Denmark/Sweden	Kattegat	Population monitoring: breeding and moult	Repeat aerial survey Annual. Breeding only monitored in Denmark.
Norway	West coast, south of 62°N	Population monitoring: Moult	Aerial survey, every 5 years
France	French North Sea and Channel coasts-Baie du Mont Saint Michel	Population monitoring: breeding and moult	Aerial surveys 18/year + 15 census (boat and land)
	French North Sea and Channel coasts-Baie de Somme and adjacent haul-outs	Population monitoring: breeding and moult	Land census every ten days (January–June). Daily from June to September
	French North Sea and Channel coasts-Baie des Veys	Population monitoring: breeding and moult	Land and aerial surveys (1/month)
Ireland	MUs to be confirmed, although 'south and southeast Ireland' and 'west Ireland' have been suggested.		

In Ireland, the harbour seal population monitoring programme incorporates moult season site monitoring, and moult season national assessment monitoring. Moult season site monitoring: each year approximately 14–16 key regional locations are subject to ground-, boat- or aerial-based estimates of maximum numbers ashore, using a standard survey protocol. At present 2–3 replicate surveys are undertaken per location per moult season. The aim is to collectively capture at least 40–50% of the "national" population in these annual monitoring surveys. Moult season national assessment: Ireland conducts a nationwide aerial thermal imaging survey for harbour seals once within each sex year Habitats Directive reporting cycle. This survey effort has thus far been undertaken in a manner identical with that in the UK, with some simultaneous ground-truthing also done in Ireland. The overall aims include estimating the minimum "national" population size, investigating changes in numbers and distribution, and providing a wider context for the annual monitoring. This work also tries to coordinate broad-scale effort with that for Northern Ireland (as in 2011–2012) thereby covering the entire island of Ireland in the same year where possible.

Table 2. Current and known plans for monitoring grey seals.

COUNTRY [13]	MAIN BREEDING AREAS MONITORED	CURRENT MONITORING	MONITORING METHOD	COMMENTS
United Kingdom	Shetland, Scotland	Pup production Monitoring	Ground count, annual since 2004	Difficult area to monitor
	Orkney, Scotland	Pup production Monitoring	Aerial survey, annual to 2010, biennial thereafter	
	Fast Castle, Isle of May and adjacent colonies, Scotland	Pup production Monitoring	Aerial survey, annual to 2010, biennial thereafter	
	Moray Firth, east Scotland	Pup production Monitoring	Aerial survey, annual to 2010, biennial thereafter	
	West Scotland	Pup production Monitoring	Aerial survey, annual to 2010, biennial thereafter	
	Western Isles, Scotland	Pup production Monitoring	Aerial survey, annual to 2010, biennial thereafter	
	Farne Islands, East England	Pup production Monitoring	Ground count, annual	
	Donna Nook and Norfolk colonies, Southeast England	Pup production Monitoring	Ground count, annual	
Netherlands	Wadden Sea	Moult and pup production	Aerial survey	Pup counts are unreliable and not appropriate to population estimates
	Delta	Extension of bird surveys	Aerial survey, monthly	No formal assessment yet. Numbers of animals in this area is thought to be low.
Germany	Schleswig-Holstein Wadden Sea	Moult and pup production	Aerial survey conducted five times per year from november to april/may; boat and land survey, annual	
Germany	Helgoland	Pup production Monitoring	Ground count, annual	

[13] In the UK, summer grey seals counts also undertaken during harbour seal surveys.

COUNTRY [13]	MAIN BREEDING AREAS MONITORED	CURRENT MONITORING	MONITORING METHOD	COMMENTS
Denmark	Limfjord	summer surveys during harbour seal moult	Repeat aerial survey; annual	
	Kattegat	Moult and pup production of Baltic population and summer surveys during harbour seal moult	aerial survey; annual	North Sea grey seals also occur in this area, and as their moult coincides with breeding of Baltic grey seals, this season is also covered, although seals from the two MUs cannot be distinguished.
Norway	Rogaland	Pup production	Ground count, every five years at least	
France	Archipelago of Molene and adjacent haul-outs	Pup production and population Monitoring	Regular (monthly) census and Photo identification	Pup counts are not appropriate to population estimates (low numbers)
	Archipelago of Sept Îles and adjacent haul-outs	Pup production and population Monitoring	Regular (monthly) Census and Photo identification	Pup counts are not appropriate to population estimates (low numbers)
	Baie de Somme and adjacent haul-outs	Pup production and population Monitoring	Regular (monthly) census and Photo identification	Pup counts are not appropriate to population estimates (low numbers)
Ireland	MUs/AUs to be confirmed, but previous monitoring has focused on 7 key areas [Sturrall, Inishkea island, Inishshark and associated islands, islands round Slyne Head, Blasket islands,Saltee Islands and Lambay island/Irelnad's Eye].		Aerial surveys on rotational basis, each of the sites surveyed once in last 4 years	

In Ireland, the grey seal population monitoring programme incorporates breeding season estimation, and summer season national assessment. *Breeding season estimation*: each year one of three regions is subject to aerial survey in order to estimate minimum pup production at its key breeding sites. The regions are: E/SE, W/SW, W/NW. The current survey regime comprises 5–6 replicates per region over the season and is designed to deliver two regional estimations per six year Habitats Directive reporting cycle. The overall aims include collectively capturing at least 80–85% of the "national" pup production and then estimating the national breeding population figure (i.e. minimum population estimate) for grey seals of all ages. *Summer season national assessment*: The nationwide aerial thermal imaging survey for harbour seals also delivers important information on grey seal summer distribution and numbers found ashore. This survey effort has thus far been undertaken alongside the harbour seal aerial thermal imaging survey in a manner identical with that in the UK. The overall aims include investigating changes in grey seal distribution and numbers at haul-out sites, and providing a wider context for the annual monitoring.

10.5 Advice on coordinated monitoring and methodology per OSPAR common MSFD indicators; Quality assurance statements

10.5.1 Seals

OSPAR contracting parties that are members of the EU should develop or sustain survey programmes to ensure that there can be a coherent assessment of the MSFD indicators for each seal AU. For grey seals in the Atlantic, it is recommended to monitor pup production (M-5), and abundance (M-3) estimated from pup count data. Total pup production should be based on models applied to replicate survey pup count data. Whereas, grey seal abundance models use a set of demographic parameters to estimate total abundance from total pup production estimates. For harbour seals, abundance (M-3) should be estimated from moulting season haul out counts. At this time during their annual cycle, harbour seals tend to spend longer at haul-out sites and the greatest and most consistent numbers of seals are found ashore. Thus the numbers presented represents the minimum number of harbour seals in each area and should be considered as an index of population size.

The following quality assurance statement is proposed in order to adequately assess trends in grey and harbour seal population size and grey seal pup production:

For grey seals, which range over considerably larger areas than harbour seals, it should be noted that depending on the parameter measured, coordination between survey cycles in different areas could be affected by the apparent variation in timing of breeding. Therefore, timing of breeding should be evaluated prior to undertaking extensive monitoring:

- undertake breeding season surveys to inform on numbers of pups born, which is the most cost-effective method for monitoring this species in the NE Atlantic;
- use pup production count data to estimate the total population size;
- undertake replicate surveys to estimate total annual pup production, and to provide confidence intervals, SEs and CVs;
- area surveyed should be consistent between years;
- for larger areas, aerial surveys are recommended rather than land-based counts;
- to determine population trends, the timing of surveys should be consistent;
- environmental covariates (e.g. state of tide, time of day and weather) should be considered.

For harbour seals:

- undertake moult surveys to provide information on population size;
- undertake replicate surveys to increase the statistical power when analysing temporal trends. (Teilmann *et al*., 2010);
- area surveyed should be consistent between years;
- for larger areas, aerial surveys are recommended rather than land-based counts;
- to determine population trends, the timing of surveys should be consistent;
- periodically replicate surveys should be performed during an extended period to assess changes in phenology (changes in the peak of the moulting season);
- environmental and other covariates (e.g. state of tide, time of day, weather and disturbances) should be considered.

Further, survey frequency and scale should be adjusted to account for changes in species range, epizootics or other significant events. Power analysis should be used to assess the effectiveness of the existing survey schemes (e.g. Meesters *et al*., 2007; Teilmann *et al*., 2010). For moult counts of harbour seals, establishing correction factors to account for seals at sea during the survey should be considered.

10.5.2 Cetaceans

Contracting parties should develop or sustain monitoring/ survey programmes to ensure that the MSFD indicators can be evaluated for each cetacean AU.

For cetaceans, it is recommended to monitor abundance “at the relevant temporal scale” for species that are regularly present (M-4), and numbers of individuals of each species being bycaught in relation to population size (M-6). Targets remain to be defined and recommendations regarding monitoring are therefore necessarily generic. Specific issues include:

a) Assessment units have been proposed (Section 10.1), but these maybe revised in the future as further evidence is gained, e.g. from ongoing genetic studies.
b) Power analyses carried out to date highlights the fact that 1% declines in abundance (as specified by the EU Habitats Directive) would be almost impossible to detect for wide-ranging cetaceans using any realistically feasible monitoring programme and, indeed, that only much larger declines are likely to be detectable in the shorter term (i.e. six yearly reporting period). The process of setting targets will need to take this into account and consider a much longer time frame (e.g. the IUCN approach of using three generations).
c) Many organisations are currently involved in cetacean monitoring in each MS. Evaluation of this monitoring has highlighted the fact that much, while clearly valuable, does not constitute a monitoring programme *per se.* In some cases the coordinated combination of regional monitoring could be sufficient to form a viable monitoring programme. In other cases, additional funding would be needed and/or methodological issues would need to be addressed.

d) Methodology for surveying abundance of most wide-ranging cetacean species is broadly similar (based on line transect surveys) and, as such, most surveys aim to collect data on all species. However, species differ in size, detectability, surface behaviour and reaction (if any) to survey vessels, so chosen methods are unlikely to be optimal for every species. Different survey methodology (e.g. photo-identification surveys) is needed for resident/ coastal cetacean populations such as bottlenose dolphins, Risso's dolphins, etc. due to their small size and localised distribution.

e) Detection of distribution range changes requires additional effort around distribution range boundaries, although distributions patterns can be extracted from line transect surveys carried out to measure abundance. Secondary data sources, such as strandings in the case of species which occur near the coast, should be integrated, to help detect range changes. Note that although a target is unlikely to be set for indicator M-2; rather changes in distribution/range are considered to be a flag for further investigation.

f) Strandings monitoring is already part of baseline cetacean monitoring for many EU countries and can provide relevant data in support of all three common cetacean indicators. Thus data on health/cause of death and life history can provide insights into population trends (M-4) and anthropogenic mortality can be diagnosed (M-6).

g) Monitoring programmes should be adaptable. For example, survey frequency and scale should be adjusted to account for changes in species range, epizootics or other significant events.

The following quality assurance statement is proposed in order to provide adequate assessment in relation to the common indicators:

10.5.3 M-4 (M-2), Wide-ranging species

For wide-ranging cetaceans, the monitoring programme will be based around large-scale abundance surveys, supported by a series of other measures:

- Large-scale dedicated abundance surveys using a combination of aerial and boat-based line transects and covering both Atlantic and Mediterranean/ Black Sea waters, including both coastal and oceanic areas in the Atlantic. (In the Mediterranean, the MSFD provision does not extend beyond 12 miles from the coast). Such surveys have taken place in the Atlantic (SCANS I and II, CODA, see (Hammond *et al.*, 2002; SCANS-II, 2008; CODA, 2009; Hammond *et al.*, 2013) but surveys are urgently needed for the Mediterranean and Black Sea. Note that this falls outside the scope of OSPAR.
- To date, such large-scale surveys have been planned on a decadal scale, reflecting the high cost. However, there would be significant benefits for monitoring power from more frequent surveys (e.g. six yearly, to coincide with Habitats Directive reporting requirements and equivalent to the IWC Revised Management Procedure). Surveys every three years are required to obtain enough power for detection of a 30% decline in three generations in species other than harbour porpoises (using α= 0.2 and CV= 0.22), and two years for detection of similar declines in porpoises (see Section 10.2). However, as outlined earlier, the degree of certainty/ precision needs to be balanced with

costs of such surveys. Based on a decadal cycle, SCANS III is now due and a proposal to the EU LIFE programme is being developed.

- To date, large-scale surveys have taken place in summer, when the weather is most suitable for observation. Large-scale dedicated winter surveys would be less cost-effective but there clearly is a need for survey data covering all seasons, in order to assess seasonal changes in distribution/abundance. Additionally, most bycatch occurs during winter fishing operations, and abundance estimates used to infer bycatch limits are obtained during summertime.
- Methods to estimate abundance from shipboard surveys during the SCANS and CODA surveys were based on standard line transect sampling (Buckland *et al.*, 1993 (Hiby and Hammond, 1989; 1993)). In particular, they followed Buckland and Turnock (1992) and Borchers *et al*. (1998); see also Hedley *et al*. (1999), Strindberg and Buckland (2004). In relation to aerial surveys, see also Hiby and Lovell (Hiby and Hammond, 1989; Hiby and Lovell, 1998). Distance methodology should be used, along with a double platform approach to estimate g(0) (i.e. the likelihood of missing cetaceans present on the transect line) and to account for any movement in the response to survey ships. See, e.g. Laake and Borchers (2004), Thomas *et al*. (2007), Faustino *et al*. (2010).
- Collection of environmental covariates (e.g. based on satellite data) will allow enhanced density surface modelling of distribution patterns within the range.
- Further work is needed to estimate g(0) from aerial surveys and including to re-examine estimation of availability bias (recent work presented at NAMMCO using data from satellite-link tags indicates that abundance estimates can vary by an order of magnitude depending on the depth at which porpoises can be detected (e.g. surface, 0.5 m, 1 m)) (Heide-Jørgensen, 2013). The proportion of time porpoises spend at the surface is likely to vary seasonally due to changes in feeding and it may therefore be unwise to assume that g(0) is always the same. Telemetry data collected concurrently with the survey could help with evaluation of g(0).
- In the inner Baltic Sea, only porpoises are present and their density is too low for line transect methodology to be effective. Alternative methodology based on Static Acoustic Monitoring has been used for estimating densities and total abundance, and producing distribution maps of harbour porpoises and identify possible hot spots, habitat preferences and areas of higher risk of conflict with anthropogenic activities (SAMBAH project[14]) and it is suggested to continue this approach (see (Gillespie *et al.*, 2005; Gillespie *et al.*, 2009; Gillespie *et al.*, 2013). Note that although within the MSFD area, this is covered by HELCOM rather than OSPAR. HELCOM has recommended that a SAMBAH-like survey should be periodically repeated e.g. every ten years (Härkönen *et al.*, 2013).

[14] www.sambah.org.

- Power to detect changes in range and abundance can be enhanced by integrating data from regional and opportunistic cetacean surveys which meet quality standards (e.g. based on the UK Joint Cetacean Protocol, e.g. Paxton *et al*., 2011). Such surveys should provide coverage both across MS waters and in all seasons of the year. More use could be made of fishery surveys (e.g. IBTS, acoustic pelagic surveys, consistent with the ICES ambition to undertake integrated ecosystem surveys; note however that cetaceans may respond to the echosounders used during acoustic surveys, and the effect of such responses on cetacean abundance estimates needs to be assessed. It is **recommended** that (a) coordinating initiatives such as JCP are extended to other MS and (b) that an adequate programme of coordinated regional and local surveys (using dedicated vessels/planes and/or platforms of opportunity such as ICES IBTS surveys or ferry routes) is supported at MS level.
- Power analyses have been published for various large and small-scale boat surveys but will be needed for all survey programmes. Any changes in methodology and/or substantial changes in species abundance/range will necessitate new power analyses.
- Strandings monitoring programmes have long formed part of the requirement to meet Habitats Directive conservation objectives as well as being recommended by ASCOBANS. In the context of the MSFD (M-2, M-4, M-6), they have an invaluable sentinel role and provide important supporting data on trends in abundance and range/distribution, in particular by identifying changes in health status, causes of mortality and life-history parameters and providing input data for population models.
- Nationally coordinated strandings monitoring programmes are needed in all MS. These should include adequate provision for notification and transport of carcases, necropsies by trained veterinary staff and processing of pathology/histopathology, contaminants, and life-history samples, as well as for analysis, population modelling and reporting. Protocols are described by Kuiken and Hartmann (1991); Geraci and Lounsbury (1993); SAC (SAC, 2000); Christensen (2001), Deaville and Jepson (2011).
- Power analyses for certain types of data derived from strandings (e.g. pregnancy rates) are available and recommendations for minimum numbers of necropsies are available (see (Murphy *et al.*, 2013).
- It would be useful to evaluate and, potentially, integrate <u>relevant</u> information arising from other kinds of monitoring and relevant research which would not otherwise be included (for example, because the data collection protocol does not meet JCP criteria), e.g. land-based (coastal) surveys, acoustic surveys (e.g. (Lewis *et al.*, 2007), monitoring of impacts of renewables developments, seismic surveys, reports of anthropogenic mortality, etc. For some species, such as sperm whales, abundance is estimated using passive acoustic monitoring (PAM).

10.5.4 M-4 (M-2), Coastal/resident populations

For coastal/resident cetacean populations which have been recognised as separate AUs (i.e. bottlenose dolphins, among the species specifically covered by the MSFD common indicators) the monitoring will comprise:

- Collection of photo-ID data routinely and regularly (including coverage of all seasons) to allow annual capture–mark recapture estimates of population size (e.g. (Wilson *et al.*, 1999; JNCC, 2005; Cheney *et al.*, 2013)). Practical guidelines for mark-recapture analysis of photo-identification data are described in Hammond (1986); Hammond (2010); Evans and Hammond (2004); Thompson *et al*. (2004); Urian *et al*. (in press).
- Power analyses have been published for various photo-ID/capture mark-recapture population estimates but are necessary for all populations. For some populations, triennial estimates are sufficient to detect abundance trends, whilst for other more frequent surveys may be required.
- Where less frequent surveys are required to detect abundance trends, consideration should be given to the value of additional data obtained through annual surveys. Such data can generate information on population dynamics, physiological health, mortality/survival rates and various reproductive parameters.
- Additional relevant data may be available from strandings monitoring and land-based surveys. Protocols for coastal and riverine sightings surveys have been published (e.g. (Dawson *et al.*, 2008).

10.6 Proposed common indicator «Blubber PCB toxicity threshold»

The UK have proposed an additional indicator at OSPAR for MSFD Descriptor 8 Contaminants. It is outlined here for information.

10.6.1 Relevance of proposed indicator

There would clearly be merit in the establishment of coordinated monitoring of a selection of pollutants in tissues of certain marine mammal species, and a subsequent coherent assessment under the MSFD. Reference can be made to the Commission Decision of 1 September 2010 (2010/477/EU), which sets out criteria to be used by the Member States to assess the extent to which good environmental status is being achieved, accompanied with references to applicable methodological standards where available.

It states that a combined assessment of the scale, distribution and intensity of the pressures and the extent, vulnerability and resilience of the different ecosystem components including where possible their mapping, allows the identification of areas where marine ecosystems have or may have been adversely affected. This approach supports the selection of the most appropriate indicators related to the criteria for assessment of progress towards good environmental status, and it facilitates the development of specific tools that can support an ecosystem-based approach to the management of human activities required to achieve good environmental status through the identification of the sources of pressures and impacts.

The indicator proposed here would fit under different descriptors proposed in the Commission Decision:

Descriptor 1 on biodiversity

One of the criteria for the assessment of species is population condition, which includes population health, fecundity rate, and survival/mortality rates (1.3.1.). Habitat condition includes chemical condition (1.6.3).

Descriptor 8 on contaminants

The concentration of contaminants in the marine environment and their effects need to be assessed taking into account the impacts and threats to the ecosystem. Member States have to consider the substances or groups of substances that (a.o.) are listed as priority substances, and/or those that may entail significant risks to the marine environment from past and present pollution. Relevant criteria are concentrations of contaminants, a.o. in biota (8.1.1.) and level of pollution effects on ecosystem components, having regard to the selected biological processes and taxonomic groups where a cause/effect relationship has been established and needs to be monitored (8.2.1).

Pertinence of indicator

Decades of rigorous experimental studies have shown that PCBs have a range of well-established dose-dependent toxic effects such as immunosuppression, endocrine disruption and reproductive impairment in all mammalian species tested, including humans (Safe, 1994). Environmental half-life for PCBs varies from a few years up to 100 years (Hickie *et al.*, 2007). Given the length of time which much of the PCB has been out in the environment, those CB congeners which are most easily degraded will have done so, and it will be mainly the more and most recalcitrant congeners which remain. Cause/effect relationships have been established for marine mammals; resulting in immunosuppression and reproductive impairment (see Section 10.6.4; (Murphy *et al.*, 2012).

Based on the proposed PCB toxicity threshold bands, many marine mammal populations in European waters are currently not achieving good environmental status (see Section 10.6.4).

10.6.2 Management activities

By the late 1990s only 1% of the globally manufactured PCBs were estimated to have reached the ocean/seawater (see Figure 10.14 reviewed in (Aguilar *et al.*, 2002).

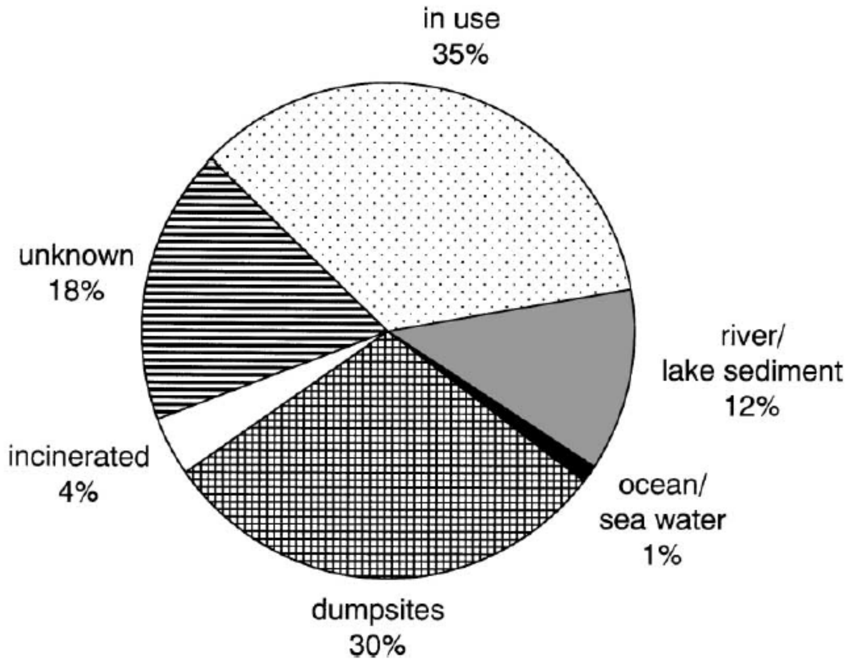


Figure 10.14. Global budget of produced PCBs kilotonnes (taken from Aguilar *et al.*, 2002).

Despite production of PCBs being stopped in Europe in 1980s and the banning of the main uses of PCBs in products in 1987 (UNEP, 2002; OSPAR, 2010), with disposal been targeted (e.g. EU directive 96/59/EC on the disposal of PCBs and polychlorinated terphenyls), there is a need for renewed steps to reduce PCB inputs into marine environment in Europe, and the continued monitoring of their toxic effects on apex predators. Potential steps for consideration include controls on disposal of buildings with PCBs added to sealants to increase durability, steps to quantify concentrations of PCBs in estuarine sediments and reduce their bioavailability to marine food chains (e.g. improved management of dredging of harbours/port) and steps to assess and mitigate sources of PCBs in waste disposal (e.g. landfill) (Jepson *et al.*, in prep.).

10.6.3 Additional information

For monitoring purposes, discussions pursued over the species to include in the assessment. It was proposed to assess the least mobile and most common, i.e. the harbour porpoises and harbour seal, which is the approach undertaken for existing biota in the OSPAR Joint Assessment and Monitoring Programme. However, the species of most concern, and those with highest rates of exposure in the OSPAR maritime area, are bottlenose dolphins and killer whales (see Section 10.6.4). If some of the populations of our main apex predators have reduced reproductive output because of high pollutants burdens, then we are <u>not</u> achieving and maintaining the good environmental status of European seas and oceans.

During the course of the WG meeting the technical specification of the proposed common PCB indicator was submitted to the OSPAR Secretariat. OSPAR HASEC was invited to discuss this proposed indicator in the context of identification of OSPAR common indicators related to MSFD D8. Additionally, this indicator was discussed at a subsequent ICG-COBAM meeting.

10.6.4 Technical specification of proposed common indicator

1) Indicator

The indicator is "Blubber PCB toxicity threshold".

2) Reasoning for the development of this indicator

Marine mammals are sentinel species for marine ecosystems (Ross, 2000). Marine mammal apex predators are vulnerable to the bioaccumulation, biomagnification and lactational transfer of specific types of lipophilic and persistent organic pollutants (POPs) such as organochlorine pesticides (e.g. DDTs and dieldrin) and industrial polychlorinated biphenyls (PCBs) (Loganathan and Kannan, 1994; Ross, 2000). In Europe, UK-stranded harbour porpoises (HPs) (*Phocoena phocoena*) showed marked and consistent declines in organochlorine pesticides, brominated flame retardants (Law *et al*., 2012a) and butyltins (Law *et al*., 2012b). Declines in the more environmentally persistent and toxic PCBs were more muted and stopped declining altogether in UK HPs around 1998 ((Law *et al*., 2012a); see Section 10.6.4; Annex 1). Species higher up marine chain, such as bottlenose dolphins (BNDs) (*Tursiops truncatus*) and killer whales (*Orcinus orca*) have even greater exposures and the highest risk of individual and population level toxicities (Law *et al*., 2012a); see Section 10.6.4; Annex 1).

Evidence base of PCB toxicity thresholds

A number of studies have proposed or established toxicity thresholds for marine mammals. One of the earliest studies established a threshold of 77 mg/kg lipid (as Clophen 50) associated with profound reproductive impairment in ringed seals (*Pusa hispida*) in the Baltic Sea (Helle *et al*., 1976). This threshold remains the highest recorded marine mammal PCB threshold (AMAP, 2004).

Experimental studies in harbour seals showed that elevated PCBs from Baltic (as compared to Atlantic) fish induced suppression of multiple indices of immunocompetence (de Swart *et al*., 1995; Ross *et al*., 1995). Blubber biopsies taken two years after the start of the experiment revealed that seals in the `contaminated' group which exhibited toxic effects had accumulated a total PCB concentration of 16.8 mg/kg lipid (or 209 ng/kg lipid total TEQ) (de Swart *et al*., 1995; Ross *et al*., 1995).

Elevated PCBs were also associated with histological indices of immunosuppression including depletion of the percentage of thymic lymphocytes in random sections of thymic tissue in UK-stranded HPs (Yap *et al*., 2012). Haematological indices of immunosuppression were also correlated with PCB exposure in a small sample of free-living BNDs in Sarasota Bay, Florida (Lahvis *et al*., 1995).

Case-control studies of mean PCBs concentrations in HPs that were "healthy" and died of acute physical trauma (control group) were compared to mean PCBs concentrations in HPs that died of a range of infectious diseases (case group). In one case-control study of UK-stranded HPs, the risk of infectious disease mortality increased by 2% for every 1% increase in summed 25CB congeners (Hall *et al*., 2006). A doubling of risk occurred at approximately 45 mg/kg lipid (blubber). In a second case-control study of UK-stranded HPs, mean summed 25 CB congeners in the 'healthy' control group (death due to physical trauma) was 13.6 mg/kg lipid, compared with 27.6 mg/kg lipid for the animals that died of infectious diseases (Jepson *et al*., 2005).

Empirical studies of BNDs in military establishments in southern California (Reddy *et al*., 2001) and free-living BNDs in Florida (Wells *et al*., 2005) have suggested that PCB concentrations around 25–30 mg/kg lipid are association with reproductive failure including abortions, stillbirths and increased first calf mortality. In the first study, OC pesticide and PCB concentrations were examined in maternal blubber prior to parturition relative to calf survivorship (Reddy *et al.*, 2001). Sum PCB concentrations for mothers with calves surviving less than twelve days were about 2.5 times those in mothers with surviving calves and many of the lost calves of mothers with high concentrations were first-borns. In the second study in Sarasota Bay, PCBs concentrations in juvenile males and females exhibited similar concentrations (15–50 mg/kg). Subsequently, males accumulated higher concentrations of PCBs through their lives (>100 mg/kg) and often died at younger age than adult females. In contrast, females begin to depurate their PCB burden with their first calf, reaching a balance between contaminant intake and lactational loss (<15 mg/kg). 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 mg/kg). Probabilistic risk assessments for reproductive impacts of PCBs on BNDs have been generated using surrogate dose-response developed from other marine mammals including otters, seals and mink (Schwacke *et al.*, 2002) and correlate well with observed effects in free-living BND populations in US (Wells *et al.*, 2005).

A PCB toxicity threshold for the onset of earliest physiological endpoints in marine mammals was determined as 17 mg/kg lipid (as Arochlor, 1254) (Kannan *et al.*, 2000). This can be considered the lowest threshold for sublethal PCB effect in exposed marine mammals and was calculated to be equivalent to 9.0 mg/kg (as sum 25 CBs (individual chlorobiphenyl congeners) lipid) (Jepson *et al.*, in prep.). The highest PCB threshold (77.0 mg/kg lipid as Clophen 50) for reproductive impairment in ringed seals in the Baltic (Helle *et al.*, 1976) was calculated to be equivalent to 41.0 mg/kg (as sum25CBs lipid) (Jepson *et al.*, in prep). Between these low and high thresholds, many PCB threshold studies suggest toxicological impacts around 25–30 mg/kg lipid in blubber.

While the toxic effects of PCBs is clearly pressure related there is a link with the state of the population (i.e. population size - indicators M-3 and M-4). Monitoring blubber PCB levels of cetaceans and seals can be considered as a key aspect in achieving GES according to the MSFD. Given the high mobility of marine mammals, and the distributional range of populations, assessments (of mean concentrations of 18/25 sum CBs) need to be made on a wide scale (population range or assessment units). Though, assessments of blubber PCB concentrations can also be undertaken at the group/cohort/individual level, which may be relevant to small populations.

3) Parameter/metric

"Total blubber PCB concentrations (as 18/25 sum CBs lipid)", determined separately for each assessment unit (AU). These AUs will vary between species. This is a European wide indicator.

4) Baseline and reference level

Various countries (e.g. UK, Germany and the Netherland) have assessed marine mammal blubber tissue PCB concentrations in historical samples. For example, in the UK blubber PCB concentrations have been assessed in stranded harbour porpoises sampled from 1990 onwards (Law *et al.*, 2012a). Historical harbour porpoise blubber samples from Germany, Denmark and Norway have been assessed for PCBs (e.g. Das *et al.*, 2006; samples collected between 1998 and 2001), and in Germany temporal variation in PCBs burdens in (historical) harbour seal samples have been investigated (Siebert *et al.*, 2012).

Baselines will vary between species/AUs. Additionally, historical tissue samples stored in European tissue banks could be assessed to establish levels over the last decade.

5) Target setting

"Blubber PCB concentrations of [marine mammal species] are reduced to below levels that are expected to allow conservation objectives to be met".

Suggested species are harbour porpoise, bottlenose dolphin, killer whale, short-beaked common dolphin, harbour seal and grey seal. Additionally, white-beaked, white-sided, Risso's and striped dolphins should be included.

The levels of PCBs in tissues are easily and accurately measured, provided blubber samples from dead stranded animals or biopsies from live animals are available and appropriate sampling and analytical methodologies are in place. Based on best scientific information available, three exposure bands are proposed:

> *Low exposure* (Good Environmental Status): 0–19 mg/kg lipid (as 18/25 sum CBs);
>
> *Moderate exposure* ("At Risk" of PCB impacts): 20–40 mg/kg lipid (as 18/25 sum CBs);
>
> *High exposure* (exceeds all published PCB thresholds"): 41+ mg/kg lipid (as 18/25 sum CBs).

Using such an approach, GES could only be considered to have been achieved when 18/25 sum CB congeners are less than 20.0 mg/kg lipid.

These PCB exposure "bands" could be applied to different groups/cohorts/individuals/populations. For example, they could be equally relevant to individuals (e.g. in small populations) or mean levels in larger populations. They could apply to all members in a population/AU (e.g. mix of males and females of all ages) or apply to specific cohorts (e.g. adult females only). Females are an important subgroup to assess PCB exposure and associated (reproductive toxicity) risk. As female cetaceans offload the majority (80%) of their PCB burden to their first born offspring during pregnancy and lactation (primarily during the first seven weeks (Cockcroft *et al*., 1989), mature females with high CB concentrations is suggestive of reproductive impairment.

6) Spatial scope

Assessment units for the relevant seals and cetaceans, also to be used in indicator M-4 and M-6 (distribution, abundance and bycatch respectively) assessments. They are, where possible, delimitated using the borders of ICES blocks as recommended by ICES WGMME (ICES WGMME, 2012) and ICES WGBYC (ICES WGBYC, 2012). Seal AUs are described in Section 10.1.

7) Monitoring requirements

The levels of PCBs in tissues are easily and accurately measured provided blubber samples from dead stranded animals or biopsies from live animals are available and appropriate sampling and analytical methodologies are in place. As part of the various European cetacean stranding programmes, cause of death, health status, nutritional condition, age and sexual maturity status of individuals are investigated. During necropsies, blubber samples are/will be collected, wrapped in foil and stored frozen (-20°C) for subsequent toxicological analysis. For offshore/small/underrepresented in strandings data marine mammal AUs, a programme of biopsying free-ranging individuals could be instigated. Along with measuring PCB concentrations in the blubber tissue of biopsied animals, genetic analysis of skin samples could be undertaken for sex determination. The full suite of 25 CBs (IUPAC numbers: 18, 28, 31, 44, 47, 49, 52, 66, 101, 105, 110, 118, 128, 138, 141, 149, 151, 153, 156, 158, 170, 180, 183, 187, 194) or 18 CBs (IUPAC numbers: 95, 101, 110 + 136, 151, 135 + 144, 149, 153, 141, 138, 187, 183 + 128, 174, 177, 171 + 202, 180, 170, 201, 203 + 196, 195, 194) will be used in assessments. Currently analysis is being undertaken to produce a correction factor for comparing datasets comprising of sum 25 CBs and sum 18 CBs.

Previously, assessment of blubber PCBs concentrations has been undertaken in various European marine mammal species inhabiting UK, Dutch, Belgium, German, Danish, French, Spanish waters, etc. as part of various national and EC funding programmes.

8) Appropriateness of the indicator

Pollution by hazardous substances, such as PCBs, is considered as one of the major anthropogenic threats to marine mammals. It is easy to understand and quantify, and there is a clear link with human activities. Analytical methods for PCB concentrations in tissues are both highly sensitive and internationally standardised for comparison with tissue PCB levels in other regions. The target set should indicate the level at which, in the absence of other important human-induced threats, conservation objectives will be met. Based on the proposed PCB exposure "bands", mean blubber PCB concentrations in bottlenose dolphins and killer whales in European waters exceed all published PCB thresholds (see Appendix 1; Jepson *et al.*, in prep.). High mean blubber PCB concentrations were also observed in European female bottlenose dolphins and killer whales, indicating reproductive impairment and/or reproductive toxicity.

Assessments will be undertaken to determine the exposure level of individuals/AUs according to the target description. It is proposed that reporting follows assessments undertaken by ICES at least every six years. This will encompass the development of a database of individual PCB pollutant levels, based on national input, which contains the relevant information from which to make such assessments.

9) Costs

Monitoring contaminant levels can be expensive. In some countries, part of the monitoring is in place (e.g. harbour porpoises in the UK), while new resources are needed for monitoring additional species.

10) Further work

- 10.1) Agreement on the AUs against which to set the target.
- 10.2) Ensure there is a standardized sample and data collection protocol for stranded animals and biopsy of free-living cetaceans in European waters.
- 10.3) A standardized reporting method and frequency needs to be developed. Agreement is needed on which body will make the assessment for AUs, although it is suggested that this should be progressed through ICES. This will encompass the development of a database of individual PCB pollutant levels, based on national input, which contains the relevant information from which to make such assessments.

10.4) Assessment of baseline and current PCB exposure in population/assessment unit levels of relevant species with greatest exposures with respect to the proposed PCB exposure "bands". Continued assessment of female offloading within each species.

10.5) Continued development of dose-response relationships between PCBs and health impacts (e.g. increased susceptibility to infectious disease mortality and reproductive impairment) for cetacean populations with consistently moderate-to-high PCB exposure (above proposed blubber PCB toxicity thresholds).

10.6.5 Literature

AMAP. 2004. AMAP Assessment 2002. Persistent Organic Pollutants (POPs) in the Arctic, Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway (2004) 293 pp.

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10.6.6 Annex 1: Additional supporting information

For harbour porpoises in UK waters, a gradual decline was initially observed in the early to-mid 1990s, followed by a "steady-state" plateau (1998–present).

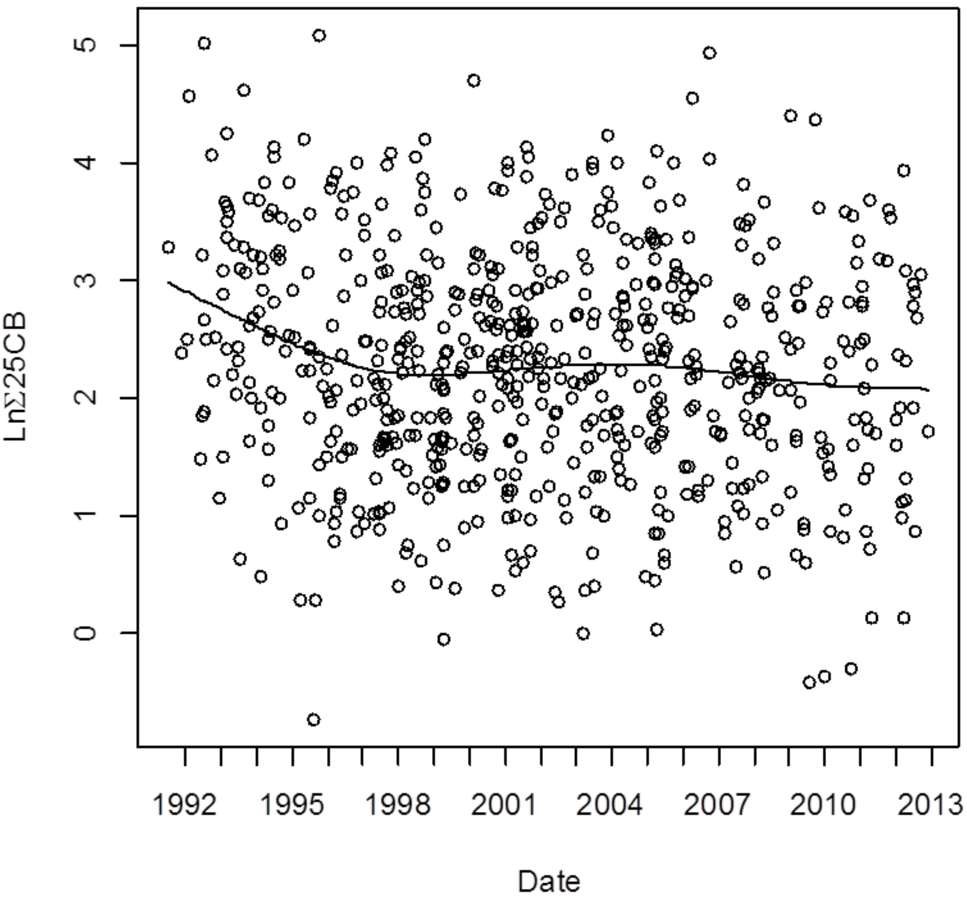
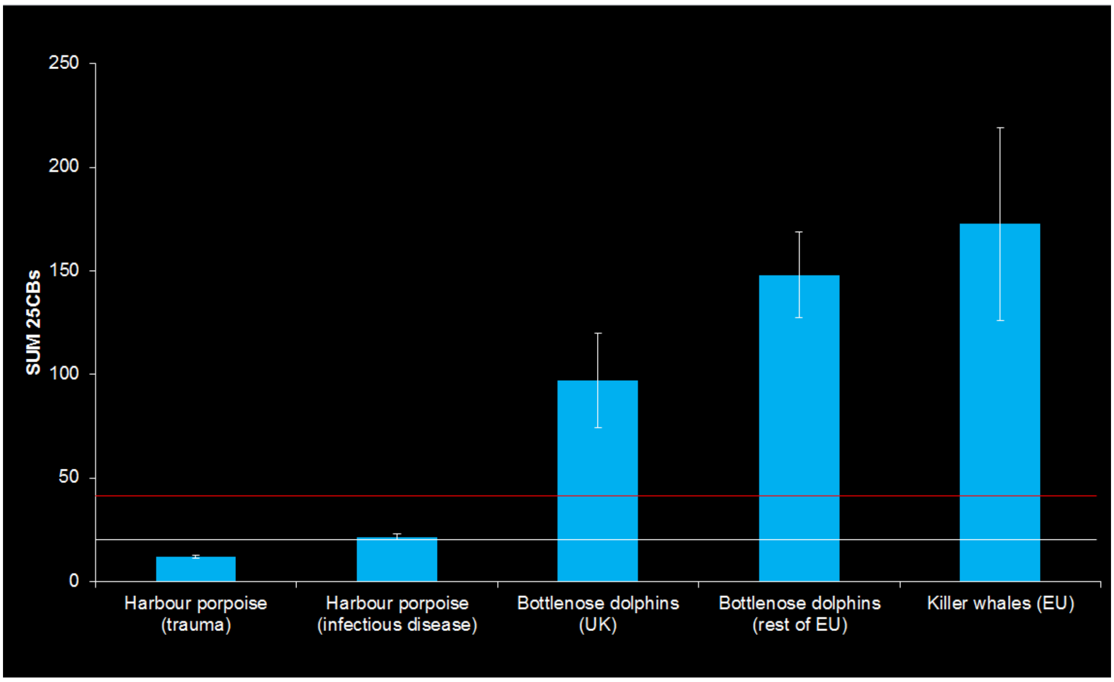


Figure 10.15. Ln P25CB concentrations in porpoise blubber against date for all data. The continuous line represents the smoothed trend from a Generalized Additive Model fitted to the data (Taken from Jepson *et al.*, in prep.)

Based on the proposed PCB exposure "bands", bottlenose dolphins and killer whales in European waters have **high levels** of exposure. High mean blubber PCB levels were also observed in female bottlenose dolphins and killer whales, indicating reproductive impairment and/or reproductive toxicity in some individuals/populations.



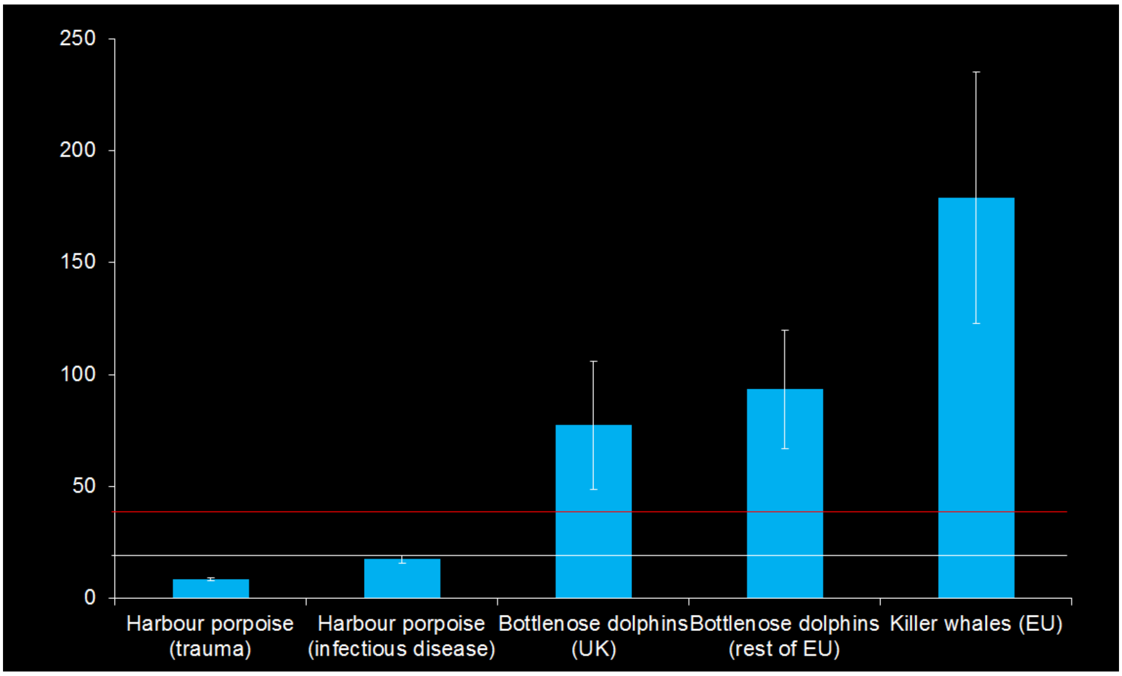


Figure 10.16. (a) Mean blubber PCBs (sum 25CBs lipid wt) species/region (*Phocoena/Tursiops/Orcinus*) (all data), (b) FEMALES only; mean blubber PCBs (sum 25CBs lipid) species/region (*Phocoena/Tursiops/Orcinus*). White line is depicting the 20 mg/kg lipid threshold, and red line is the 41 mg/kg lipid threshold (Taken from Jepson *et al.*, in prep.). PCB exposure bands: low exposure (Good Environmental Status): 0–19 mg/kg lipid (as 18/25 sum CBs); moderate exposure ("At Risk" of PCB impacts): 20–40 mg/kg lipid (as 18/25 sum CBs); high exposure (exceeds all published PCB thresholds"): 41+ mg/kg lipid (as 18/25 sum CBs). (Taken from Jepson *et al.*, in prep.).

10.7 Recommendations

WGMME **recommends** that the OSPAR consider the proposals made in relation to their request. In summary these are:

1) Adoption of the proposed assessment units for grey and harbour seals in the OSPAR Maritime area; assessment units for the more common cetacean species are also proposed.

2) Focus monitoring and assessment on M3 and M4 (trends in abundance), with M1 and M2 (range and pattern of distribution) being removed from the list of common indicators; and subsumed within M3 and M4, respectively. It is not possible to propose a firm and measurable baseline, metric and target for common indicators M1 or M2. Distribution changes should act as warning signals and research should be carried out to investigate the causes of those changes, especially to determine if they have an anthropogenic cause.

3) Consider the technical and scientific advice provided on options for setting targets, determining baselines and associated monitoring requirements. Some standards for monitoring are suggested.

4) With respect to M6 (bycatch), it is not possible to progress this indicator significantly. Further work and collaboration between the European commission, ICES and OSPAR is required. WGMME have recommended that the European Commission give serious consideration to ICESs offer to host a workshop, with the objective of reviewing different mechanisms for determining safe bycatch limits and finalising conservation objectives for a bycatch limit approach that would enable conservation aspiration to be met.

5) OSPAR ICG-COBAM and HASEC should continue evaluating the proposed common PCB indicator for use within Descriptors 1 and 8, respectively.

The WGMME recommends that collaboration with ICG-COBAM is maintained for the continued development of the mammal common indicators.

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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 likely 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 6 of this document.

Recommendation I

WGMME strongly supports the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2016 and **recommends** that it is supported by all range states and by ICES, ASCOBANS and the European Commission. Continuation of these surveys is essential to the accurate estimation of absolute abundance for several species that are required for reporting under the Habitats Directive and the Marine Strategy Framework Directive.

Recommendation II

As part of the reforms to the Common Fisheries Policy and the Data Collection Framework, the European Commission requested that ICES provide advice on the use of management frameworks and other mechanisms for determining safe bycatch limits in 2013. The ICES response noted that further work in this area would be required and that: '*This could be in the form of a workshop for invited participants representing managers, scientists and stakeholders. As stressed in the advice, input from management and from the "societal" side is crucial for such a process. We would envisage attendees from relevant parts of the European Commission (at least DG Mare and DG Environment), Member State fisheries authorities, the RACs, relevant intergovernmental bodies (Regional Seas Commissions, ASCOBANS and ACCOBAMS) and relevant NGOs.* Given that the lack of agreed conservation objectives is the primary reason stopping further development, WGMME **recommends** that European Commission give serious consideration to ICESs offer to host a workshop, with the objective of reviewing different mechanisms for determining safe bycatch limits and finalising conservation objectives for a bycatch limit approach that would enable conservation aspiration to be met.

Recommendation III

WGMME **strongly recommends** that ICES members of the OSPAR region provide data so that the seal database can be maintained and updated regularly. Such development is considered essential to future MSFD assessments of the OSPAR core set of indicators for seals.

Recommendation IV

There is a wide range of monitoring methodologies available to assess marine mammals at marine renewable energy development sites, but not all techniques are equally appropriate to all sites. Moreover, assessing the suitability of techniques and the quality of resulting survey data can be hampered by incomplete reporting of methodological details by developers. Commercial sensitivities may further complicate efforts by regulators and others to compare monitoring techniques on their respective merits.

WGMME **recommends** that regulators and policymakers should require the use of open, transparent and reproducible survey and monitoring methodologies to assess potential impacts on marine mammals. Furthermore, for line-transect surveys, the data should be fit to provide absolute densities. For all monitoring, the use of established and peer-reviewed methods is encouraged, acknowledging that new innovations or methods may arise. Methods associated with such new techniques should be sufficiently well described so that conclusions arising from these techniques are reproducible. Data from surveys should be made publicly available in formats that allow future reanalysis (for example using JCP-type protocols).

Recommendation V

WGMME **recommends** that the OSPAR consider the proposals made in relation to their request. In summary these are:

1) Adoption of the proposed assessment units for grey and harbour seals in the OSPAR Maritime area; assessment units for the more common cetacean species are also proposed.
2) Focus monitoring and assessment on M3 and M4 (trends in abundance), with M1 and M2 (range and pattern of distribution) being removed from the list of common indicators; and subsumed within M3 and M4, respectively. It is not possible to propose a firm and measurable baseline, metric and target for common indicators M1 or M2. Distribution changes should act as warning signals and research should be carried out to investigate the causes of those changes, especially to determine if they have an anthropogenic cause.
3) Consider the technical and scientific advice provided on options for setting targets, determining baselines and associated monitoring requirements. Some standards for monitoring are suggested.
4) With respect to M6 (bycatch), it is not possible to progress this indicator significantly. Further work and collaboration between the European commission, ICES and OSPAR is required. WGMME have recommended that the European Commission give serious consideration to ICESs offer to host a workshop, with the objective of reviewing different mechanisms for determining safe bycatch limits and finalising conservation objectives for a bycatch limit approach that would enable conservation aspiration to be met.
5) OSPAR ICG-COBAM and HASEC should continue evaluating the proposed common PCB indicator for use within Descriptors 1 and 8, respectively.

The WGMME recommends that collaboration with ICG-COBAM is maintained for the continued development of the mammal common indicators.

Annex 1: Participants list

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Annex 2: Agenda

ICES WGMME 2014: Marine Research Facility, Woods Hole Oceanographic Institution, Quissett Campus, USA

Monday, 10th March 2014

09:30 Welcome and start of meeting

Fred Serchuk (US Delegate to ICES and an ICES Vice President): Introduction to the ICES Strategic Plan

Michael Moore (Director - WHOI Marine Mammal Center): Overview of the Woods Hole Oceanographic Institution (WHOI), US Geological Survey (USGS) and Marine Biological Laboratory (MBL) programmes at Woods Hole

Michael Simpkins (Chief, PSB) - Overview of Protected Species Branch (PSB) Marine Mammal programme

11:30 Plenary session: setting up of Internet connections, adoption of agenda; Forming of subgroups and leads, setting up of work plan

12.30 Lunch break

13:45 Work in subgroups

15.30 Coffee break

15.45 Work in subgroups

18:30: Close

Tuesday, 11th March 2014 (link with Oban meeting, UK)

08:30 Start (confirm numbers for Friday's fieldtrip and visits to Aquarium)

08:45 (12:45 UK time) Presentations by Chris Orphanides and Marjorie Lyssikatos on bycatch estimation for harbour porpoise & white sided dolphin

09:45 (13:30 UK time) Presentation by Allison Henry on Large Whale Serious Injury Determination

10:30 (14:00 UK time) Presentation by Tim Cole on Right Whale Surveys

11.00 (14:30 UK time) Plenary session: update from leads of ToRs

11:30 Coffee break

11:45 Work in subgroups

13.30 Lunch break

15:00 Work in subgroups

16.30 Coffee break

16.45 Work in subgroups

18:30 Close

Wednesday, 12th March 2014 (link with Oban meeting, UK)

08:30 Start

08:45 (12:34 UK time) Presentation by Richard Pace: Right Whale Population Estimates

09:30 (13:30 UK time) Presentation by Beth Josephson on AMAPPS database

10:00 (14:00 UK time) Plenary session; review of material from ToR F (Oban meeting) and any other completed ToRs

11:00 Coffee break

11:30 Work in subgroups

13:30 Lunch break

15:00 Work in subgroups

16:30 Coffee break

16:45 Plenary session: review print outs of available first drafts

18:30 Close

19:00 Dinner (optional) place to be announced

Thursday, 13th April 2014 (link with Oban meeting, UK)

08:30 Start

08:45 (12:45 UK time) Presentation by Dani Cholewiak: Using Passive Acoustics to Estimate Abundance

09:30 Subgroup discussion - bycatch frameworks.

11:00 Coffee break

11:30 Work in subgroups; finalizing reports

12:00 Plenary session: review of finalised material for each ToR

13:00 Lunch break

14:00 Plenary session: review of finalised material for each ToR

16:30 Coffee break

16:45 Plenary session: review of finalised material for each ToR

18:30 Close of meeting

Annex 3: PART C: Technical specification of proposed common biodiversity indicators

Mammals

Code	Previous code*	Indicator	Category
M-1	31&33	Distributional range and pattern of grey and harbour seal breeding and haul-out sites, respectively	Core
M-2	32&34	Distributional range and pattern of cetaceans species regularly present	Core
M-3	35	Abundance of grey and harbour seal at breeding and haul-out sites, respectyively	Core
M-4	36	Abundance at the relevant temporal scale of cetacean species regularly present	Core
M-5	37	Grey seal pup production	Core
M-6	38&39	Numbers of individuals within species being bycaught in relation to population	Core

Draft OSPAR Common Indicators: marine mammals (M-2)

Distributional range and pattern of cetacean species regularly present

1) Indicator

"Distributional range and distributional pattern within range of cetacean species regularly present".

The cetacean species for use as a core indicator under OSPAR are limited to the following species:

- harbour porpoise
- bottlenose dolphin
- white-beaked dolphin
- minke whale
- common dolphin

Common dolphin are considered representative of the wider European waters (i.e. both off and on the continental shelf). It should also be noted that bottlenose dolphins can be divided into two types. There are well known small resident coastal groups (possibly to be divided into different management units) and groups, comprising much more animals, that are wide-ranging both inshore and offshore ('oceanic' population).

2) Reasoning for the development of this indicator

Marine mammals, including cetaceans, are top predators, and comprise an important part of biodiversity (Descriptor 1). As all cetacean species are taken up under the Habitats Directive (annex IV and/or II), their distribution comprises a key aspect for securing and achieving GES according to the MSFD.

With the possible exception of some coastal bottlenose dolphin populations, cetaceans are generally mobile over large spatial and temporal scales. For example, there was a significant southerly shift in the North Sea harbour porpoise population between the two SCANS surveys (1994 and 2005). Assessments therefore need to be undertaken at an appropriate scale and it should be noted that expansions in range are far easier to detect than contractions. A good understanding of natural movement patterns (e.g. seasonal patterns) is required prior to any deterioration or expansion being detected and links made with anthropogenic activities.

Because of the scale required for assessments, a transboundary approach to the collection, collation and analysis of data will be required. Such an approach has also been suggested for Favourable Conservation Status assessments for the Habitats Directive.

Number of CPs reporting/using the indicator (n=9): 8

Consensus among CPs on usefulness as part of a region wide set (n=8): 8

3) Parameter/metric

"Distributional range of cetacean species regularly present and distributional pattern at the relevant temporal scale of cetacean species regularly present."

There is a very clear overlap between distributional range and distributional pattern within range. The same monitoring will be used to undertake both analyses. An assessment of distribution, including trends over time, is required as part of the Favourable Conservation Status (FCS) assessments for the Habitats Directive (as short-term and long-term trends) [15].

4) Baseline and reference level

Although the baseline should be based on historical data, these are not available at the appropriate spatial and temporal scale. Moreover, the historical distributional range and pattern of many cetacean species cannot realistically be restored (assuming it has contracted, which is unknown for many species) as today's marine environment is very different. Climatic changes may have important consequences. For the harbour porpoise, there have been important distributional shifts in the North Sea during the last decades. For the coastal bottlenose dolphin, many populations are small, and some estuaries that historically contained populations no longer do so (e.g. Humber an Thames Estuaries, UK); in other locations (e.g. the Sado Estuary, Portugal), populations are endangered. The relationship between inshore and 'oceanic' populations is not well known, and the much larger 'oceanic' populations are relatively poorly known.

White-beaked dolphins occur over a large part of the European continental shelf, including the North Sea, but are rare in the Irish Sea, Celtic Sea, Channel and Bay of Biscay, and around the Iberian Peninsula.

Minke whales are widely distributed in European shelf waters, particularly along the Atlantic seaboard and in the northern and central North Sea.

For common dolphins, there are large seasonal movements in the population on and off the continental shelf, whilst in some areas the possibility of 'inshore' and 'off-

[15] In the 2007 FCS assessments, this was undertaken on a country by country basis which led to an unsatisfactory standard of assessment at the European North Atlantic scale (ICES, 2009). For the 2013 FCS assessments, a greater emphasis has been placed on the need for a transboundary approach (European Commission, 2011), although it seems unlikely that this will occur.

shore' populations has been suggested. For this species, as with bottlenose dolphin, it is essential that assessments include consideration of the species off the continental shelf.

5) Target setting

The proposed target is "Maintain populations in a healthy state, with no decrease in population distribution with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state". Some difficulties can be encountered here, because there is usually no straightforward link between the distributional range and pattern, and human activities. Although the baseline for each species considered should be based on historical data, these are generally not available at the appropriate spatial and temporal scale.

6) Spatial scope

The geographical scope of the indicator is species dependent. With the exception of coastal bottlenose dolphin populations, cetacean populations cover large spatial scales often extending beyond European North Atlantic waters for example. Assessments therefore need to be undertaken at an appropriate scale and a good understanding of natural variability and patterns of movement is required prior to any change of distribution being detected and links made with anthropogenic activities. Management Units for cetacean species, also to be used in indicator M-4 (Abundance) and M-6 (bycatch) assessments, have been loosely defined by ASCOBANS (Evans and Teilmann, 2009), reviewed by WGMME (2012) and further refined by WGMME (2013; see Appendix 1).

7) Monitoring requirements

The objective of the monitoring should be to detect trends, in particular negative ones, in the distributional range and pattern, due to human pressures. Human pressures are diverse: some human activities remove individuals directly from the population (e.g. bycatch). Other pressures degrade condition and health of animals (e.g. contaminants, food depletion), or displace populations towards habitats of poorer quality (disturbance by noise, habitat modification). Monitoring is undertaken through a variety of approaches and by many different organisations. There are large-scale international surveys such as SCANS and CODA, annual national surveys that occur in the waters of some Member States and, at a more localised scale, and there are various surveys undertaken by the state, academic institutions and/or non-governmental organisations. Although these surveys are mostly dedicated to provide for density estimation, they also yield information about distribution and distributional patterns.

Strandings data represent to date the most extensive and long-term source of demographic data for a number of cetacean populations (at least in areas where strandings occur). Strandings data are currently clearly underexploited and rarely analysed at an international level. They could yield useful complementary information to identify possible anthropogenic impacts, and can contribute to the identification of possibly underlying reasons for trends in the distributional range and pattern of cetaceans. Coverage needs to be reliable, and biological and pathological investigations need to be standardised.

The monitoring and assessment undertaken for distributional range and pattern of cetaceans, will be made in combination with indicator M-4 (abundance).

8) Appropriateness of the indicator

In most cases it is difficult to find a straightforward link between the range and the distribution pattern of cetaceans and human activities. There are multiple pressures, and climate change is also a factor influencing abundance and distribution. However, as top predators and being charismatic animals of general public concern, changes in distribution and abundance are important, and should be assessed against changes in human activities and climate change to detect cause–effect relationships and, where necessary followed by the appropriate management measures.

9) Reporting

Given that populations have a transboundary distribution (except for the resident and most coastal bottlenose dolphins groups) agreements have to be made on monitoring and reporting in order to be able to make an assessment. The reporting frequency should follow the monitoring frequency, and the assessment for most species should be made every six years. For the small cetaceans it is proposed that ICES makes the assessment, while for the minke whale a regular assessment of the Northeast Atlantic population is made by the IWC.

10) Costs

Monitoring distribution and distributional range of cetacean can range from fairly cheap (monitoring of an inshore population with a limited range) to very expensive (monitoring of an offshore population distributed over a large area); however, part of the monitoring is in place (in a combination of indicator M-2, M-4 and M-6), while new resources are needed, e.g. for large-scale decadal surveys and more comprehensive annual surveillance (see also indicator M-4).

11) Further work

Future steps are similar as for the indicator M-4 (abundance).

11.1) Compilation of existing data on the distributional range;

11.2) Development of a baseline for each species;

11.3) Development of a method to extract data on distribution and distributional pattern from the data obtained from the monitoring for indicator M-4;

11.4) Development of, and agreement on, a standardized reporting and assessment method;

11.5) For small cetaceans, agreement on the body that provides for the assessments.

Literature

European Commission. 2011. Assessment and reporting under Article 17 of the Habitats Directive Explanatory Notes & Guidelines for the period 2007–2012. Available at: http://circa.europa.eu/Public/irc/env/monnat/library?l=/habitats_reporting/reporting_2007–2012/reporting_guidelines&vm=detailed&sb=Title.

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Thomas, L. 2009. Potential Use of Joint Cetacean Protocol Data for Determining Changes in Species' Range and Abundance: Exploratory Analysis of Southern Irish Sea Data. Available at: http://jncc.defra.gov.uk/pdf/JCP_Prelim_Analysis.pdf.

Appendix 1: Management Units

WGMME (2013) recommended that Member States use the following management units for reporting requirements of the Habitats Directive and for the development of indicators and their assessment for the Marine Strategy Framework Directive.

There is a single MU in the European North Atlantic for common dolphin (*Delphinus delphis*), white-beaked dolphin (*Lagenorhynchus albirostris*), white-sided dolphin (*Lagenorhynchus acutus*), striped dolphin (*Stenella coeruleoalba*) and minke whale (*Balaenoptera acutorostrata*).

For harbour porpoise (*Phocoena phocoena*), MUs are proposed for the Iberian Peninsula, Celtic Sea, Irish Sea, West Scotland/NW Ireland, the North Sea and Inner Danish waters (Figure 1). More than one MU in the North Sea for harbour porpoise should be explored in ongoing work to develop management models for setting safe limits to bycatch.

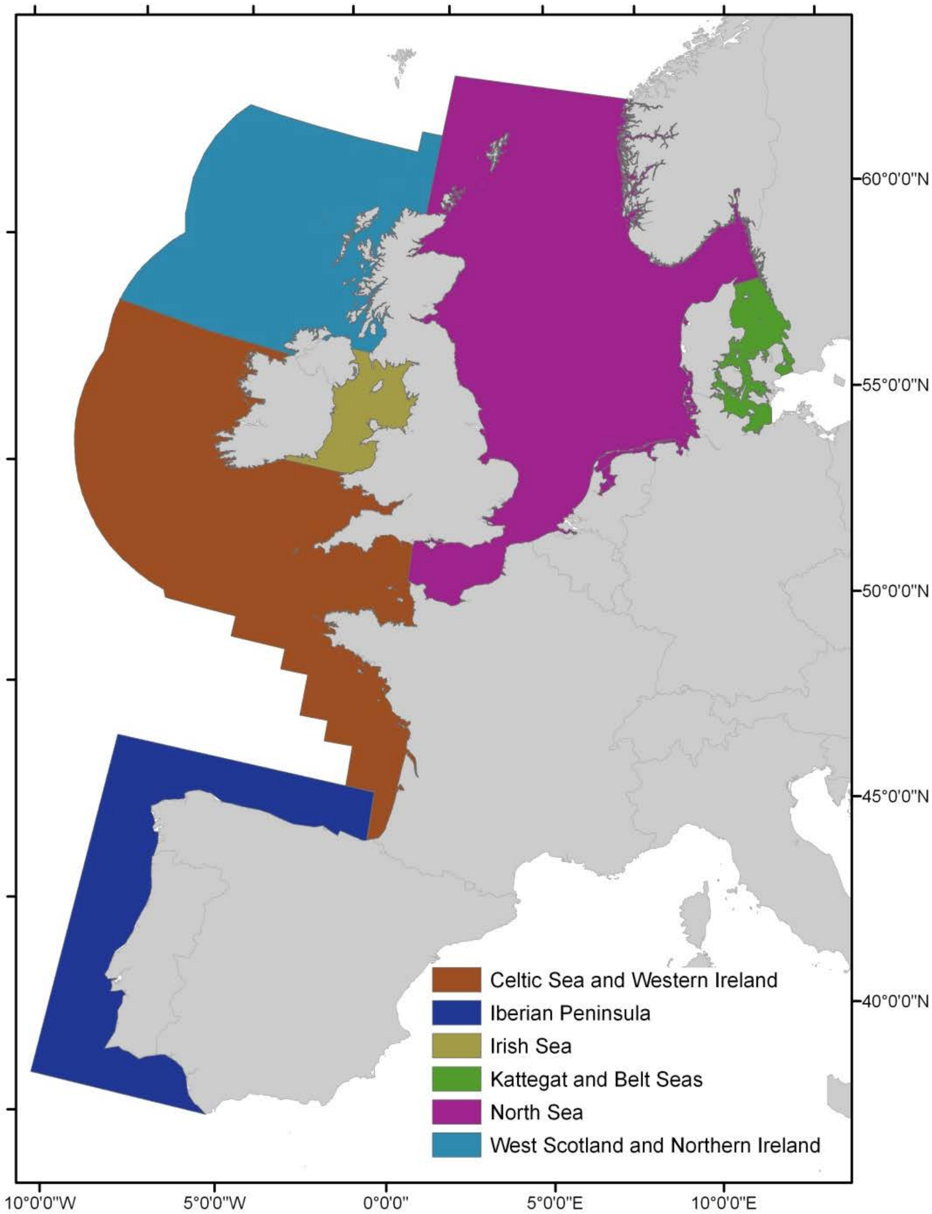

Figure 1. Harbour porpoise management units.

Bottlenose dolphins have a complex population structure, with three types being recognized: very small residential groups, slightly wider ranging resident coastal groups and the oceanic group. The following management units are proposed (given from north to south; Figure 2):

Resident groups:_Barra (Scotland); Shannon Estuary (Ireland); Ile de Sein and Archipel de Molene (France); Galician rias (Spain); Sado Estuary (Portugal).

Coastal groups: east coast of Scotland (UK); Inner Hebrides (UK); Irish Sea (Ireland and UK); Connemara-Mayo (north and west coasts of Ireland); south coast of Ireland; the coastal English Channel/Celtic Sea (UK); north coast of France, north coast of Spain; Galicia (Spain); coast of Portugal; the Azores (Portugal), Gulf of Cadiz (south coast of Spain) and Strait of Gibraltar (south coast of Spain).

Oceanic waters: a single MU for all continental shelf/slopes/oceanic waters outside 12 nm from the coast. It should be noted that although this MU extends into the North Sea (represented by ICES Area IV, excluding coastal east Scotland) and that very few bottlenose dolphin are seen in this area and, although there is no conclusive evidence, those seen are thought to belong to the Coastal Scottish group.

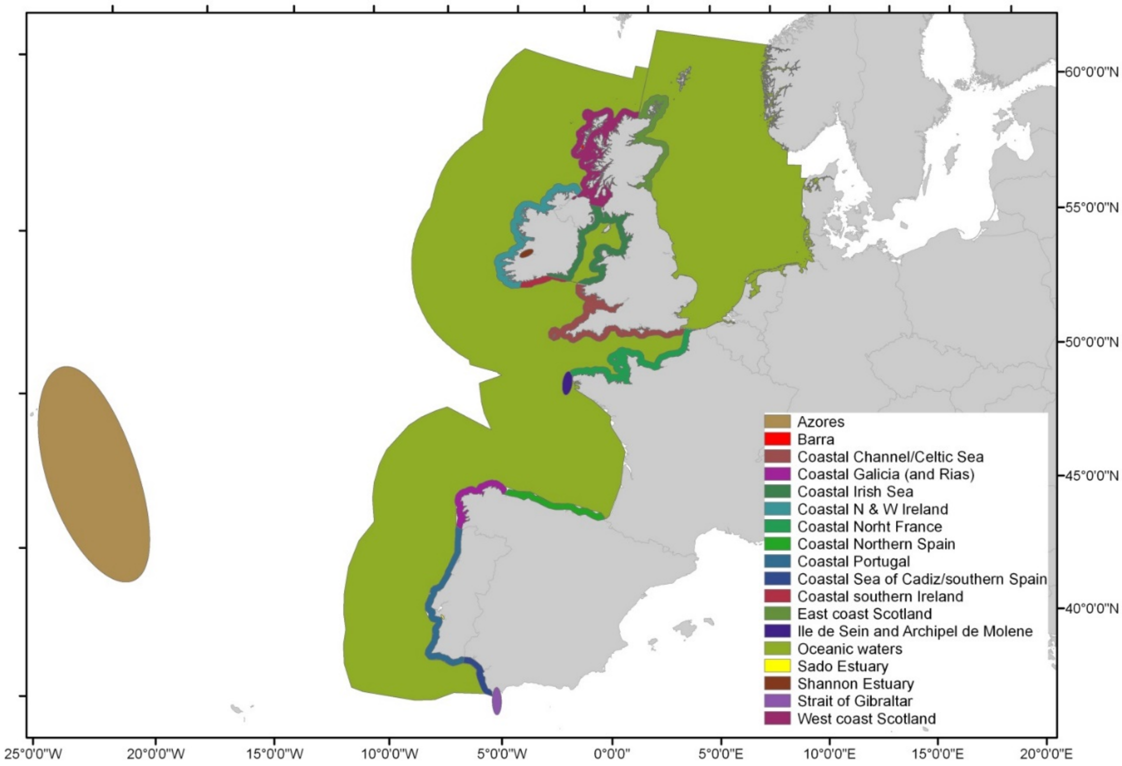

Figure 2. Bottlenose dolphin management units.

PART C: Technical specification of proposed common biodiversity indicators

Mammals

CODE	PREVIOUS CODE*	INDICATOR	CATEGORY
M-1	31&33	Distributional range and pattern of grey and harbour seal haul-outs and breeding colonies	Core
M-2	32&34	Distributional range and pattern of cetaceans species regularly present	Core
M-3	35	Abundance of grey and harbour seal at haul-out sites & within breeding colonies	Core
M-4	36	Abundance at the relevant temporal scale of cetacean species regularly present	Core
M-5	37	Harbour seal and Grey seal pup production	Core
M-6	38&39	Numbers of individuals within species being bycaught in relation to population	Core

Draft OSPAR Common Indicators: marine mammals (M-3)

Abundance of harbour and grey seals at haul-out sites & within breeding colonies

1) Indicator

"Abundance of harbour and grey seals".

2) Reasoning for the development of this indicator

Marine mammals, including seals, are top predators, and comprise an important part of biodiversity (Descriptor 1). As harbour and grey seal are taken up under the Habitats Directive (annex II), their abundance comprises a key aspect for securing and achieving GES according to the MSFD.

Number of CPs reporting/using the indicator (n=9): 7

Consensus among CPs on usefulness as part of a region wide set (n=8): 7

3) Parameter/metric

"Abundance, at the appropriate spatial and temporal scale, of harbour and grey seal at haul-out sites and/or within breeding colonies (as appropriate)".

Existing OSPAR EcoQO's encompass grey seal pup production (which is scaled up to provide abundance estimates) and the population size of harbour seals (estimated from haul-out counts), but the monitoring for this indicator would also yield necessary information for indicator M-1 (distributional range and pattern) and M-6 (bycatch). In practice, for harbour seals the counts at haul-out sites and colonies are used to establish trends and changes, although these only comprise a part of the true population; at some locations estimates of true abundance can be made by using information of the proportion of the animals counted. For grey seals, estimates of the population size are made through a method similar as for harbour seals or through a population model fitted to pup production.

4) Baseline and reference level

Although the baseline should derive from historical data, these are not available everywhere. Moreover, the historical abundance of seals at haul-out sites and colonies is a situation that cannot realistically be restored, given for instance large-scale coastal developments and tourism. Climatic changes and outbreaks of PDV may have important consequences. It is therefore likely that a modern baseline will have to be used, such as a favourable reference situation for abundance at the different Management Units (MUs), as defined in the Favourable Conservation Status assessments under the Habitats Directive or maximum counts in the last decade. However, as different countries have set different baselines, there might be a need for a more coherent definition. Baselines could be set to the level at which population growth rates are levelling off due to natural causes, with a need to decide a time period over which this is measured.

5) Target setting

The proposed target is: "Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state".

This target should be set for every Management Unit (MU). MUs should not be specifically listed in the target (as is the case now in the OSPAR EcoQO), thus avoiding the need to rewrite or update the wording of the indicator as new information on populations comes to light. A restoration will not be feasible if anthropogenic activities have increased to a level where habitats are no longer suitable. Identifying trends in colonies near the edge of the range of harbour and grey seals will be especially important, as will movements of seals between MUs (immigration and emigration).

6) Spatial scope

For monitoring the EcoQO's on seals, the North Sea has been subdivided into different monitoring areas. A subdivision of the European populations into MUs is proposed for the whole range of both species at Annex 1. While population estimates are made at the MU level through combining site level estimates, movements between MUs need to be taken into account.

7) Monitoring requirements

It is possible to detect changes in abundance of harbour seals from haul-out counts and for grey seals from pup counts and, where this is not possible, from moult counts. In most parts of the distributional range of the harbour and grey seal, there is sufficient monitoring at haul-out sites and/or breeding colonies. This monitoring also provides information for the parameters M-1 (distributional range and pattern), M-5 (pup production of grey seal) and M-6 (bycatch). However, some MUs are monitored annually, whereas UK and Norwegian harbour seal MUs are monitored annually up to every fifth year.

In the Wadden Sea, the monitoring and management under the Trilateral Monitoring and Assessment Programme and Wadden Sea Plan (Trilateral Seal Agreement; CMS) are well established over the last decades, and support the indicators and targets for harbour seals, and (although not under CMS) also the ones for grey seals. Similar work has also been ongoing in the UK over a similar time frame.

Monitoring methods: any method that yields abundance estimates per MU.

Monitoring frequency: different per MU, but at least once every five years.

8) Appropriateness of the indicator

Although no straightforward link exists between the abundance of seals and human activities, a number of human activities may lie at the basis of trends and changes in abundance. The monitoring of the indicator serves as to trigger the investigation of possible cause–effect relationships as a basis for measures. Changes due to epizootics might be important. For example, Phocine Distemper Virus (PDV) has caused past declines in European harbour seal populations, with the first and most significant outbreak in 1988 and the second in 2002. Also, there have been recent increases in the grey seal populations, and climate change may have important consequences for both species.

9) Reporting

Given that most populations have a transboundary distribution, and that shifts between colonies and haul-out sites can occur, agreements have to be made on monitoring and reporting in order to be able to make an assessment. The reporting frequency should be in line with the monitoring frequency.

As ICES has made overviews of the abundance of seals at different Management Units in the past, it is suggested that data are sent to ICES for assessment in the frame of the implementation of the MSFD.

Assessments should be made at least every six years.

10) Costs

Costs should be relatively low, given that seal colonies are inshore. The monitoring should be combined with the monitoring for indicators M-1 (distributional range and pattern) and M-5 (pup production), and will serve an assessment of M-6 (bycatch).

11) Further work

Future steps are similar for the indicators M-1 (distributional range and pattern) and M-5 (pup production).

11.1) Agreement is needed on the management units proposed in Annex 1.

11.2) Existing data for an agreed time period within each MU need to be compiled.

11.3) Baselines need to be set for each MU.

11.4) A standardized reporting method and frequency needs to be developed together with an assessment method; agreement is needed on which body will make the assessment.

Literature

OSPAR. 2009. Evaluation of the OSPAR system of Ecological Quality Objectives for the North Sea (update 2010). OSPAR Biodiversity Series, 406.

PART C: Technical specification of proposed common biodiversity indicators

Mammals

CODE	PREVIOUS CODE*	INDICATOR	CATEGORY
M-1	31&33	Distributional range and pattern of grey and harbour seal haul-outs and breeding colonies	Core
M-2	32&34	Distributional range and pattern of cetaceans species regularly present	Core
M-3	35	Abundance of grey and harbour seal at haul-out sites & within breeding colonies	Core
M-4	36	Abundance at the relevant temporal scale of cetacean species regularly present	Core
M-5	37	Harbour seal and Grey seal pup production	Core
M-6	38&39	Numbers of individuals within species being bycaught in relation to population	Core

Draft OSPAR Common Indicators: marine mammals (M-4)

Abundance at the relevant temporal scale of cetacean species regularly present

1) Indicator

"Abundance, at the relevant temporal scale, of cetacean species regularly present".

The cetacean species for use as a core indicator under OSPAR are limited to the following species:

- harbour porpoise
- bottlenose dolphin
- white-beaked dolphin
- minke whale
- common dolphin

Common dolphin are considered representative of the wider European waters (i.e. both off and on the continental shelf). It should also be noted that bottlenose dolphins can be divided into two types. There are well known small resident coastal groups (possibly to be divided into different management units) and groups, comprising much more animals, that are wide ranging both inshore and offshore ('oceanic' population).

2) Reasoning for the development of this indicator

Marine mammals, including cetaceans, are top predators, and comprise an important part of biodiversity (Descriptor 1). As cetaceans are taken up under the Habitats Directive (Annex IV), their abundance (criterion 1.2.) comprises a key aspect for securing and achieving GES according to the MSFD. However, as it is not feasible to monitor all cetaceans, which include uncommon, widely dispersed and oceanic species, the indicator is limited to the population size of management units (MUs) of a number of shelf species for which objectives were set or measures proposed in the framework of OSPAR, ASCOBANS, EC fishery regulations and the Habitats Directive (Annex II).

The monitoring and assessment of the indicator is partly in place, with monitoring already required under the Habitats Directive and fisheries legislation (Regulation 812/2004 and Data Collection Regulation).

Number of CPs reporting/using the indicator (n=9): 8

Consensus among CPs on usefulness as part of a region wide set (n=8): 8

3) Parameter/metric

"Abundance of cetacean species regularly present (at the relevant temporal and spatial scale)".

The same monitoring used to assess changes in cetacean abundance will be used to assess changes in distribution (M-2). An assessment of abundance, including trends over time, is required as part of the Favourable Conservation Status (FSC) assessments for the Habitats Directive[16].

4) Baseline and reference level

Although the baseline should derive from historical (i.e. pre-1900) data, these are not available at the appropriate spatial and temporal scale. Moreover, the historical abundance of many cetacean species is unknown and cannot realistically be restored (where it is known to have declined) as today's marine environment is very different. Climatic changes may have important consequences. A modern baseline has to be utilised for the species considered. However, abundance estimates typically have wide confidence values, and may not have the power to detect a statistically significant trend. Therefore, abundance data should always be considered with any available data on distributional changes, causes of death in stranded animals and possible links with human activities.

5) Target setting

A general proposed target for all species is: "Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state".

For coastal bottlenose dolphins it could be further refined to: "*Maintenance of the current levels of the populations where stable, and where feasible and relevant, an increase in numbers*". A recovery in areas where it was known to occur up to the 20th century might not be realistic in the short or medium term, given the life-history parameters of bottlenose dolphins, with a slow reproduction. However, as several of the estuaries they occupied in the past are now much cleaner than they were, and fish are returning to them (e.g. Thames and Clyde estuaries), it is possible that they return to colonise these areas within a few decades.

For cetacean populations with a relatively small range, FCS could also be used.

6) Spatial scope

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[16] In the 2007 FCS assessments, this was undertaken on a country by country basis which led to an unsatisfactory standard of assessment at the European North Atlantic scale (ICES, 2009). For the 2013 FCS assessments, a greater emphasis has been placed on the need for a transboundary approach (European Commission, 2011), although this is unlikely to occur.

The geographical scope of the indicator is species dependent. With the exception of some coastal bottlenose dolphins, cetacean populations cover large spatial scales often extending beyond European North Atlantic waters for example. Assessments therefore need to be undertaken at an appropriate scale and a good understanding of natural variability and patterns of movement is required prior to any decline or increase in population size being detected and links made with anthropogenic activities. MUs for cetacean species, also to be used in indicator M-2 (distributional range and pattern) and M-6 (bycatch) assessments, have been defined by ASCOBANS (Evans and Teilmann, 2009), were further reviewed by ICES (2012), and were further adapted by ICES (2013) to, where possible, take account of well-known ICES block boundaries, specifically for bycatch assessment. MUs for all relevant species are proposed in Annex 1. Reference/baseline levels for each MU of:

- Harbour porpoise and white-beaked dolphin: can be derived from large-scale surveys (SCANS);
- Common dolphin and bottlenose dolphin (wide ranging oceanic populations): can be derived from large-scale surveys (SCANS, CODA);
- Bottlenose dolphin (coastal populations): can be derived from mostly long-term local/regional photo-ID studies;
- Minke whale: can be taken from the regular surveys undertaken by TNASS and Norwegian surveys, with additional information from large-scale surveys (SCANS, CODA); IWC undertakes regular assessments.

7) Monitoring requirements

The abundance of cetaceans can be monitored using a variety of techniques. Because of the scale required for assessments, a transboundary approach to the techniques used, and the collection, collation and analysis of data will be required. Also strandings data can be useful as complementary information to identify possible anthropogenic impacts[17], and can contribute to the identification of possible underlying reasons for trends.

The objective of the monitoring should be to detect trends, in particular negative ones, in the abundance of cetacean populations due to human pressures. As cetacean monitoring is costly, the frequency at which data should be collected shall depend on the species monitored; it can be yearly and with a high resolution for species with a limited range (e.g. for coastal bottlenose dolphin) up to decadal and with a coarse resolution for wide ranging species. Monitoring is undertaken through a variety of approaches and involves many different organisations. There have been large-scale international surveys such as SCANS and CODA, annual national surveys in the waters of some Member States and, at a more localised scale, various surveys under-

[17] Strandings data represent to date the most extensive and long-term source of demographic data for a number of cetacean populations (at least in areas where strandings occur). Although they cannot yield an absolute figure for abundance, they can in some cases be interpreted to provide for a relative indication of local and temporal variations in coastal abundance. Strandings data are currently clearly underexploited and rarely analysed at an international level. In addition, the investigation of stranded cetaceans can yield information on a number of life-history parameters, and on causes of death, and therefore provide some indications about human pressures.

taken by the state, by academic institutions and/or non-governmental organisations[18]. For the monitoring of this indicator, a coordinated combination of these types of survey will be required.

Since part of the monitoring is used to set baselines against which to set bycatch limits or trends, boundaries for MUs were defined (Annex 1), where possible taking account of well-known ICES block boundaries.

Monitoring methods and frequency for:

- Harbour porpoise: aerial- and ship-based surveys; large-scale surveys every six years, complemented by more frequent surveys at a smaller spatial scale that yield information on a higher spatial and temporal resolution; such surveys also yield information for white-beaked dolphin;
- Common dolphin and bottlenose dolphin (wide ranging 'oceanic' populations): aerial- and ship-based surveys; large-scale surveys every six years;
- Bottlenose dolphin (coastal populations): photo-ID as the main method, but ship-based surveys can be appropriate; annually;
- Minke whale: regular surveys undertaken by TNASS and Norwegian surveys, with additional information from large-scale surveys (SCANS, CODA).

8) Appropriateness of the indicator

There is usually no straightforward link between the abundance of cetaceans and human activities. There are multiple pressures, and climate change is an additional factor influencing abundance and distribution. However, as top predators and animals general public concern, changes in distribution and abundance are important, and should be assessed against changes in human activities and climate change to detect cause–effect relationships, where necessary followed by the appropriate measures.

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[18] A mechanism, the Joint Cetacean Protocol, is being developed that can bring these disparate datasets together at the NW European Atlantic scale (JCP, Paxton *et al.*, 2011, see http://jncc.defra.gov.uk/page-5657). Effort-related cetacean sightings data from all major data sources are included e.g. SCANS I & II, CODA, European Seabirds at Sea (ESAS), SeaWatch Foundation (SWF) and other non-governmental organisations, as well as industry (e.g. in relation to potential renewable energy installations in UK waters). These data, collected between 1979 and 2010, represent the largest NW European cetacean sightings resource ever collated. It is recognised, however, that there are some significant datasets missing such as the annual national monitoring undertaken by some States. It is expected that the JCP will deliver information on the distribution, relative abundance and population trends of the more regularly occurring cetacean species occurring in NW European waters. A preliminary phase of the project, covering the Irish Sea and west coast of Scotland, was recently been completed (Paxton *et al.*, 2011). This work was used to refine the modelling techniques that had been developed in earlier projects (Thomas, 2009; Paxton and Thomas, 2010; Paxton *et al.*, 2011). A final analysis of northwest European waters will be published in 2013.

9) Reporting

Given that populations have a transboundary distribution (except for some coastal bottlenose dolphins), agreements have to be made on monitoring frequency. The reporting frequency should follow the monitoring frequency, and the assessment for most species should be made at least every six years. For the small cetaceans it is proposed that ICES makes the assessment, while for the minke whale a regular assessment of the Northeast Atlantic population is made by the IWC.

10) Costs

Cetacean monitoring can range from fairly cheap (monitoring of an inshore population with a limited range) to very expensive (monitoring of an offshore population distributed over a large area). Part of the monitoring is in place (in a combination of indicator M-2, M-4 and M-6), while new resources are needed, e.g. for annual surveillance and large-scale decadal surveys (see also indicator M-2).

11) Further work

Work has begun on several subjects, but further work and/or agreement is needed:

11.1) A compilation of existing data on abundance.

11.2) An agreement on the delimitation of MUs; a proposal is made at Annex 1.

11.3) The development of a baseline for each species in each MU.

11.4) The development of a standardized monitoring methodology, or alternatively a mechanism for standardizing data post collection. Although progress has been made, both effort-related monitoring of cetaceans and analytical procedures need further refinement and standardisation.

11.5) For small cetaceans, the development of an assessment tool and agreement on the body that makes the assessment.

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PART C: Technical specification of proposed common biodiversity indicators

Mammals

CODE	PREVIOUS CODE*	INDICATOR	CATEGORY
M-1	31&33	Distributional range and pattern of grey and harbour seal haul-outs and breeding colonies	Core
M-2	32&34	Distributional range and pattern of cetaceans species regularly present	Core
M-3	35	Abundance of grey and harbour seal at haul-out sites & within breeding colonies	Core
M-4	36	Abundance at the relevant temporal scale of cetacean species regularly present	Core
M-5	37	Harbour seal and Grey seal pup production	Core
M-6	38&39	Numbers of individuals within species being bycaught in relation to population	Core

Draft OSPAR Common Indicators: marine mammals (M-5)

Harbour seal and Grey seal pup production

General remark by the working group: pup production is in most MUs for harbour seals very difficult to monitor, and in practice is only undertaken at one MU (Wadden Sea). Although ICES acknowledges the value of such monitoring, the indicator cannot be considered as a *common indicator* that should be monitored at every MU, as it is technically not possible to establish such monitoring at the majority of MUs. Therefore ICES advises to only use grey seal pup production as a common indicator to be used in the implementation of the MSFD. If this would be accepted, the text below should be adapted where appropriate.

1) Indicator

"Fecundity rate of [harbour seal and] grey seal (pup production)".

2) Reasoning for the development of this indicator

Marine mammals, including harbour and grey seals, are top predators, and comprise an important part of biodiversity (Descriptor 1). As harbour and grey seal are taken up under the Habitats Directive (Annex II), their population condition comprises a key aspect for securing and achieving GES according to the MSFD.

Grey seals form breeding aggregations at traditional, remote colonies, with females often returning to the same location on the breeding colony to give birth to their single pups. In addition, some females exhibit philopatry; i.e. returning to breed at their natal site. It is for these reasons that some grey seal population estimates are based on pup counts. In contrast, harbour seals do not aggregate into discrete colonies to breed. The females appear to move away from larger groups to give birth and raise their new-born pups in very small groups, returning to form larger groups when the pup is sufficiently old. The dispersed nature of the breeding groups and the fact that pups are able to swim within hours of birth contrive to make estimating pup production of harbour seals extremely difficult in some areas. It is for this reason that population estimates for harbour seals are undertaken during their annual moult when groups tend to be larger than at other times of the year and numbers at many haul-out sites appear to be at a maximum. However, in some areas (the Wadden Sea and limited rocky shore areas such as the Kalmarsund in Sweden), counts are made during the breeding season for harbour seals.

Number of CPs reporting/using the indicator (n=9): 7

Consensus among CPs on usefulness as part of a region wide set (n=8): 7

3) Parameter/metric

"[Harbour seal and] grey seal pup production in each Management Unit"

4) Baseline and reference level

Although the baseline should derive from historical data, these are not available everywhere. Moreover, the historical distributional range of breeding sites and colonies is a situation that cannot realistically be restored, given for instance coastal developments and tourism. Climatic changes may have important consequences. It is therefore likely that a modern baseline will have to be utilized, such as average pup production in the last decade per MU.

5) Target setting

The target is "No statistically significant long-term average decline of ≥10% at each management unit".

While an existing OSPAR EcoQO deals with grey seal pup production, there is not an equivalent to harbour seal pup production.

6) Spatial scope

A proposal for the delimitation of MUs is presented in Annex 1.

7) Monitoring requirements

The monitoring required takes place in combination with the monitoring for the indicators M-1 (distributional range and pattern) and M-3 (abundance). There is sufficient monitoring at breeding colonies for grey seals. In contrast, for harbour seals it will not be possible to cover all MUs, as it is much more difficult to count harbour seal pups. Harbour seal counts are undertaken during the breeding season in the Wadden Sea. In the Wadden Sea, the monitoring and management under the Trilateral Monitoring and Assessment Programme and Wadden Sea Plan (Trilateral Seal Agreement; CMS) are well established since a few decades, and support the indicators and targets for harbour seals, and (although not under CMS) also the ones for grey seals. The monitoring of grey seal pup production takes place every year or every two years at all MUs.

8) Appropriateness of the indicator

There is usually no straightforward link between a human activity and pup production. There are multiple pressures, such as disturbance, coastal engineering works and pollution, possibly affecting pup production, or causing spatial shifts of pup production over time. However, changes and trends are important to detect cause–effect relationships between pup production and a certain human activity, where necessary to be followed by appropriate measures.

9) Reporting

Given that some MUs are transboundary, and that shifts may occur between adjacent colonies, agreements have to be made on the appropriate time-scale of monitoring and reporting in order to be able to make an assessment. It is proposed that the reporting frequency follows the monitoring frequency. It is further proposed that data are submitted to ICES for an assessment at least every six years.

10) Costs

As the monitoring is coastal in nature, costs are limited; the monitoring can be combined with the monitoring for the indicators M-1 (distributional range and pattern) and M-3 (abundance).

11) Further work

Future steps are similar for the indicators M-1 (distributional range and pattern) and M-3 (abundance).

11.1) Compilation of existing data on pup counts.

11.2) Agreement on the proposal in MUs as in Annex 1.

11.3) Development of a baseline for each MU (where possible).

11.4) Development of a standardized monitoring methodology, or alternatively, a mechanism for standardizing data post collection.

11.5) Agreement on a time-scale for monitoring and reporting, development of an assessment tool and agreement on the body that makes the assessment.

Literature

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PART C: Technical specification of proposed common biodiversity indicators

Mammals

CODE	PREVIOUS CODE*	INDICATOR	CATEGORY
M-1	31&33	Distributional range and pattern of grey and harbour seal breeding and haul-out sites, respectively	Core
M-2	32&34	Distributional range and pattern of cetaceans species regularly present	Core
M-3	35	Abundance of grey and harbour seal at breeding nad haul-out sites, respectively	Core
M-4	36	Abundance at the relevant temporal scale of cetacean species regularly present	Core
M-5	37	Grey seal pup production	Core
M-6	38&39	Numbers of individuals within species being bycaught in relation to population	Core

Draft OSPAR Common Indicators: marine mammals (M-6)

Mortality of seals and cetaceans due to bycatch

1) Indicator

The indicator is “mortality due to bycatch”.

2) Reasoning for the development of this indicator

Marine mammals are usually slowly reproducing, and a high human-induced mortality, on top of natural mortality, can have serious and long-term implications for the population. An important source of human induced mortality that can be singled out is bycatch in fishing gear. While the number of animals bycaught is clearly pressure related, there is a link with a state of the population (population size - indicators M-3 and M-4).

For cetaceans, the Habitats Directive requires that incidental capture or killing is monitored, and that it should not have a significant negative impact on the species. Therefore the setting of limits for bycatch of cetaceans can be considered as a key aspect in achieving GES according to the MSFD. It has been agreed that bycatch targets can also be set for pinnipeds, as bycatch also occurs in these marine mammals. As the maximum population growth rates differ in marine mammals, different targets will be needed. Given the high mobility of marine mammals, and the distributional range of populations, assessments will necessarily need to be made on a wide scale (population range or management units). Difficulties exist in both measuring bycatch and population size in a sufficiently high degree of accuracy to draw conclusions, and in combining data originating in different regions for an overall assessment of GES.

Number of CPs reporting/using the indicator (n=9): 7

Consensus among CPs on usefulness as part of a region wide set (n=8): 7

3) Parameter/metric

"*Numbers of individuals being bycaught in relation to population size estimates*", determined separately for each management unit (MU). These MUs will vary between species.

4) Baseline and reference level

Although some historical bycatch estimates exist, the current levels of bycatch in relation to the population estimates (baseline), and a trend-based target can be used.

5) Target setting

The target "The annual bycatch rate of [marine mammal species] is reduced to below levels that are expected to allow conservation objectives to be met" may require different approaches for different species. Although bycatch occurs in a wide range of species, it should only be specifically assessed for those species for which there is sufficient data. Suggested species are harbour seal, grey seal, harbour porpoise, short-beaked common dolphin and striped dolphin. However, noting the occurrence of bycatch in other species may be useful information when assessing the factors possibly affecting the abundance and distribution (considered in M-2 and M-4). Although some targets have been proposed and accepted, a review of these is currently being made. New targets will be proposed for each relevant species and for each relevant MU.

The harbour porpoise bycatch limit reference point of 1.7% is derived from work undertaken by a working group convened by the International Whaling Commission and ASCOBANS (IWC, 2000). This has subsequently become the standard target or level above which bycatch is considered to be unsustainable. However, there has been much debate about the use of a simple fraction of the best population estimate. A very simple deterministic population dynamics model was used, which assumed a "biological" population with independent population dynamics. If this management target is to be applied to management regions for harbour porpoise, the animals living in the areas defined by these regions are assumed to have more or less independent dynamics (which is clearly not the case in the European North Atlantic). Where the population dynamics are not independent, the management targets calculated on the basis of biological populations are unlikely to be appropriate. An alternative to such an approach is the bycatch management procedures developed under the SCANS-II and CODA projects (Winship, 2009).

In 2009, ICES advised the European Commission 'that a Catch Limit Algorithm approach is the most appropriate method to set limits on the bycatch of harbour porpoises or common dolphins. In order to use this (or any other) approach, specific conservation objectives must first be specified. In both species improved information on bycatch and the biology of the species would improve the procedure.' In 2010, ICES again advised the European Commission that 'ICES advised in 2009 of the need for explicit conservation and management objectives for managing interactions between fisheries and marine mammal populations. This advice has not been acted upon. Lacking these objectives, ICES is unable to properly consider the impacts of these interactions in its management advice.' WGMME (2013) noted again that this advice still had not been acted upon and, to aid such decisions, suggested that ASCOBANS be asked to consider the policy decisions required for the setting of safe bycatch limits.

An alternative for the parameter (bycatch as a proportion of the population size) is the use of the current bycatch numbers as the baseline and aim for it to be reduced in future years. This would mean that no information is required on the population size, but have the significant disadvantage that there is no link with the population state. Using such an approach, GES could only be considered to have been achieved when there was no longer any bycatch.

6) Spatial scope

Management units (MUs) for the relevant cetaceans, also to be used in indicator M-2 and M-4 (distribution and abundance) assessments, are proposed in appendix. They are, where possible, delimitated using the borders of ICES blocks as recommended by WGMME (2012) and WGBYC (2012). Seal MUs still need to be clearly defined.

7) Monitoring requirements

In 2008, the International Council for the Exploration of the Sea (ICES) Working Group on Marine Mammal Ecology tried to evaluate progress to date with the harbour porpoise bycatch EcoQO on a North Sea wide basis (WGMME, 2008). 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 Regulation 812/2004. Subsequently, ICES Working Group on Bycatch of Protected Species has tried to evaluate the impact of fisheries bycatch annually.

Extrapolated estimates of total bycatch in EU waters in 2009 (based on EC/812/2004 national reports) were available for striped dolphins (about 870), for common dolphins (around 1500), for bottlenose dolphins (ten) and for harbour porpoises (about 1100) (WGBYC 2011). It is clear that these totals provide only a very patchy overview of total cetacean bycatches within European waters due to low and uneven sampling coverage (WGBYC, 2011). Reductions in bycatch should be considered as a target that will contribute to GES, but it is currently not possible to evaluate whether the indicator will provide an accurate assessment of GES. However, data collation techniques are continually improving and coverage of the relevant fisheries sectors has been increasing.

Problems in monitoring are the scale of assessment (marine mammal population distributions are wider than national waters), monitoring of bycatch is undertaken using different methodologies and to different standards, and, in some Member States, bycatch can occur in the recreational or part-time fishery sector, which is considerably harder to monitor.

As part of their national developments of MSFD indicators and targets, the UK is following ICES advice and has started work on the use of management frameworks for determining safe limits to bycatch for harbour porpoise, short-beaked common dolphin, bottlenose dolphin, harbour seal and grey seal. This work, however, is not being restricted to national waters.

A source of information, currently underexploited, are strandings. These not only provide demographic data for cetacean populations, but can also be used to detect changes in the causes of death within some degree of confidence, certainly with species for which sufficient numbers wash ashore (WGBYC, 2012; WGMME, 2012). Although absolute estimates should be treated with caution, trends are likely to be informative, and a good coverage and a standardised methodology is needed.

8) Appropriateness of the indicator

Bycatch is considered as one of the major anthropogenic threats to marine mammals. It is easy to understand and quantify (although the methods for quantification are not straightforward), and there is a clear link with human activities (different fishing métiers). The target set should indicate the level at which, in the absence of other important human-induced threats, conservation objectives will be met.

9) Reporting

The proposed target means that knowledge is required both on bycatch and on the population size, both spatially and temporally, and within appropriate confidence values. This poses problems, as has been demonstrated by WGBYC (2010). With the available data on bycatch of harbour porpoises it was not possible to conclude whether or not the set target had been met during the most recent years. Estimates of bycatch were made on the basis of the number of fishing days per fisherman, the landings in relevant fisheries, and on board observer schemes. Currently, observer schemes are not required in all relevant fisheries according to the fisheries legislation. There is an obligation under the Habitats Directive to monitor bycatch, but it has to date not been enforced by the European Commission, and obligations also exist under the Common Fisheries Policy.

It is proposed that reporting follows the monitoring, and that the assessment of the bycatch of seals and small cetaceans is undertaken by ICES at least every six years. WGBYC have developed a database of bycatch based on national reports which contains the relevant information from which to make such assessments.

10) Costs

Both monitoring marine mammal abundance (indicators M-3 and M-4) and bycatch rates can be expensive, especially where a high coverage of fisheries through independent observers on board is required. Cheaper methods exist, such as the use of camera systems on board, or a voluntary reporting scheme by fishermen.

11) Further work

There is clearly a lack of information on aspects of this indicator, although information is slowly improving. Concerning the population sizes of the marine mammals, and the assessment scale, the lack of information and proposed future steps are described in the summaries of the indicators M-3 and M-4 (Abundance). Concerning bycatch, the following aspects should be further developed through linkages with appropriate fora:

11.1) Agreement on the MUs against which to set the targets; a proposal for cetaceans is included in Appendix 1.

11.2) Development of safe bycatch limits for each species and MU.

11.3) A standardized reporting method and frequency needs to be developed together with an assessment tool. Agreement is needed on which body will make the assessment, although it is suggested that this should be progressed through ICES.

11.4) Investigation of the use of stranded animals to derive information on trends in causes of mortality.

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Appendix 1: Management Units

WGMME (2013) recommended that Member States use the following management units for reporting requirements of the Habitats Directive and for the development of indicators and their assessment for the Marine Strategy Framework Directive.

There is a single MU in the European North Atlantic for common dolphin (*Delphinus delphis*), white-beaked dolphin (*Lagenorhynchus albirostris*), white-sided dolphin (*Lagenorhynchus acutus*), striped dolphin (*Stenella coeruleoalba*) and minke whale (*Balaenoptera acutorostrata*).

For harbour porpoise (*Phocoena phocoena*), MUs are proposed for the Iberian Peninsula, Celtic Sea, Irish Sea, West Scotland/NW Ireland, the North Sea and Inner Danish Waters (Figure 1). More than one MU in the North Sea for harbour porpoise should be explored in ongoing work to develop management models for setting safe limits to bycatch.

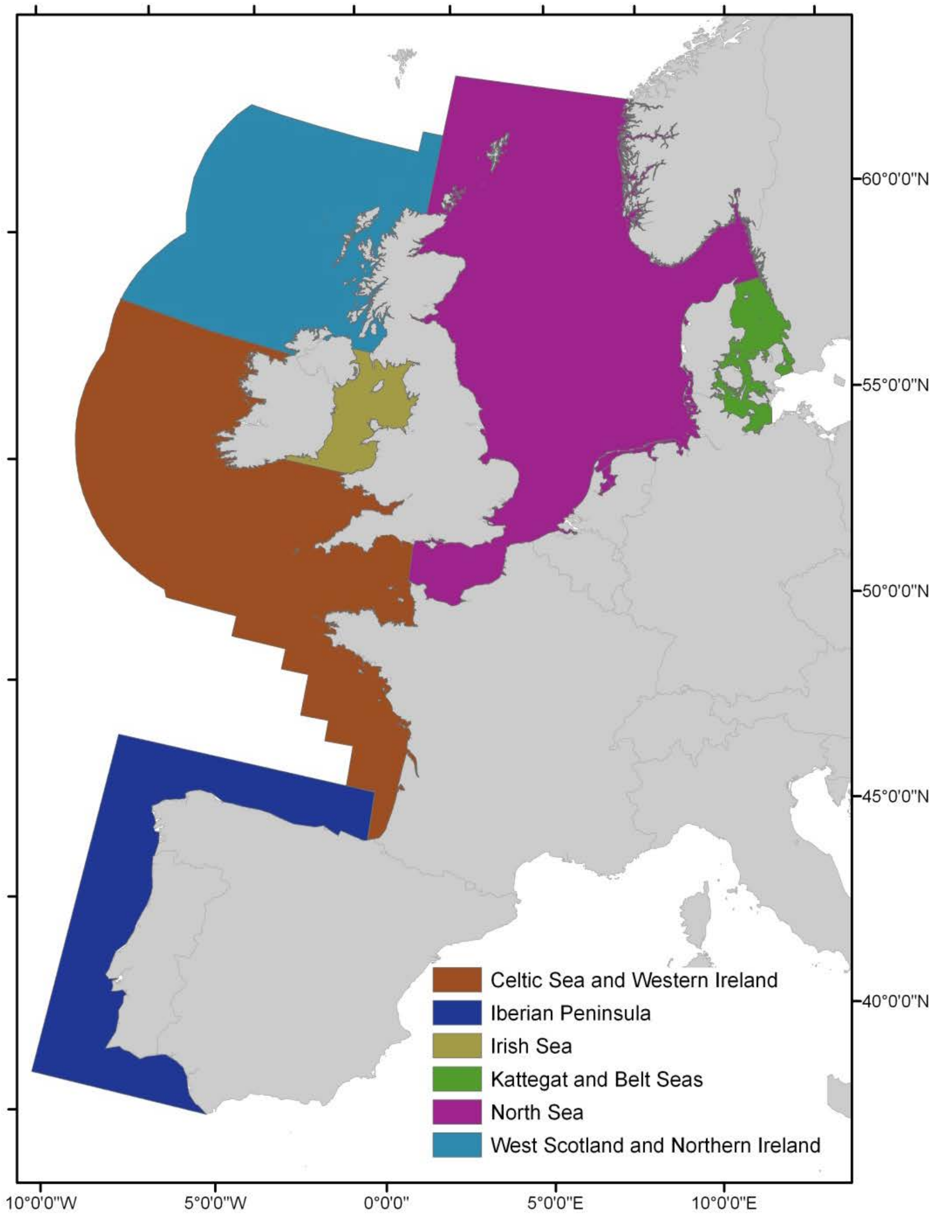

Figure 1. Harbour porpoise management units.

Bottlenose dolphins have a complex population structure, with three types being recognized: very small residential groups, slightly wider ranging resident coastal groups and the oceanic group. The following Management Units are proposed (given from north to south; Figure 2):

Resident groups:_Barra (Scotland); Shannon Estuary (Ireland); Ile de Sein and Archipel de Molene (France); Galician rias (Spain); Sado Estuary (Portugal).

Coastal groups: east coast of Scotland (UK); Inner Hebrides (UK); Irish Sea (Ireland and UK); Connemara-Mayo (north and west coasts of Ireland); south coast of Ireland; the coastal English Channel/Celtic Sea (UK); north coast of France, north coast of Spain; Galicia (Spain); coast of Portugal; the Azores (Portugal), Gulf of Cadiz (south coast of Spain) and Strait of Gibraltar (south coast of Spain).

Oceanic waters: a single MU for all continental shelf/slopes/oceanic waters outside 12 nm from the coast. It should be noted that although this MU extends into the North Sea (represented by ICES Area IV, excluding coastal east Scotland) and that very few bottlenose dolphin are seen in this area and, although there is no conclusive evidence, those seen are thought to belong to the Coastal Scottish group.

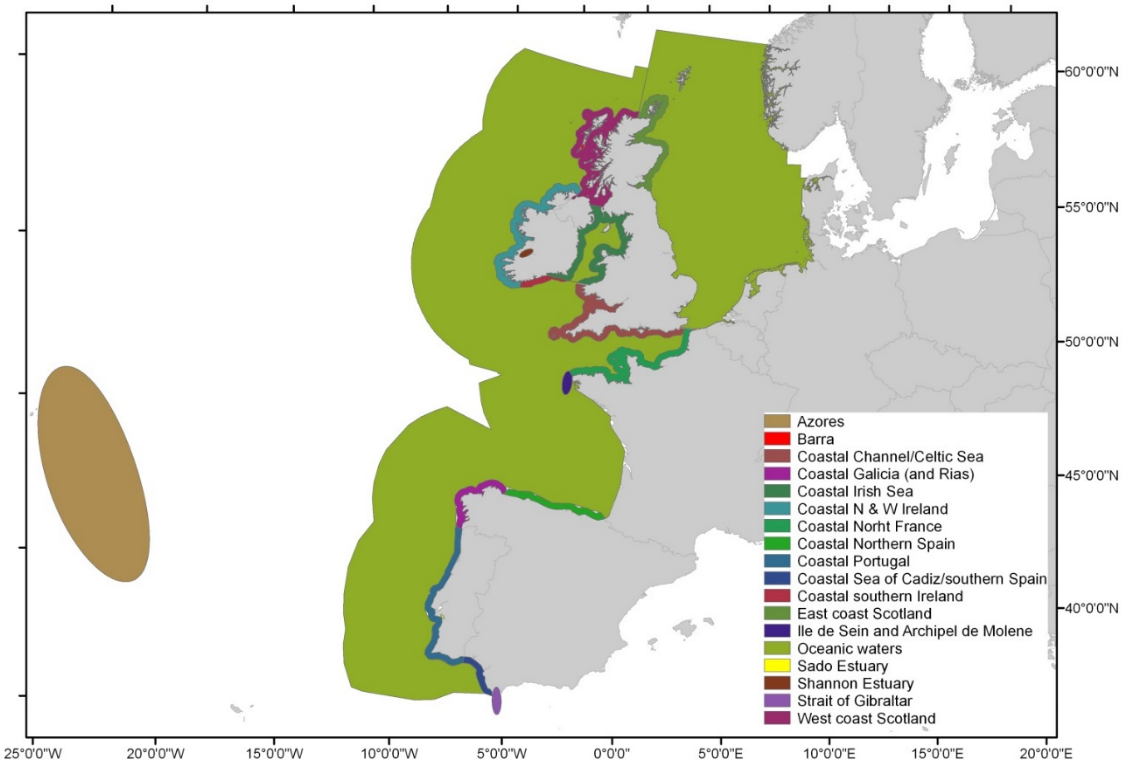

Figure 2. Bottlenose dolphin management units.

Annex 4: Technical Minutes from the Marine Mammals Review Group

- RGMME
- Review deadline 16 April
- Peer Reviewers: Garry Stenson (Canada) R1; Olle Karlsson (Sweden) R2; Finn Larsen (Denmark) R3 and Claus Hagebro from the ICES Secretariat
- Working Group: WGMME

General comments

R1

The Working Group has done a significant amount of work pulling together a tremendous amount of information on a diversity of topics in order to provide the advice they have. They are to be congratulated. Clearly this report builds upon previous reports, many of which I am not familiar with. Therefore, it was hard, at times, to follow (especially given all of the acronyms) and I may have missed some things and/or repeat the obvious. Also, some of my comments may be inappropriate given the history of these requests. I am not quite sure exactly what is to be provided for the advice as some sections seem to address some issues while other sections address others. However, I think most of my comments are fairly minor.

R2

In general I agree with the recommendations given by the expert group, the advice is relevant as well as the review of interactions between aquaculture and marine mammals. Using marine mammals as biodiversity indicators are complicated, historical data are often missing or anecdotal and many populations have been exploited heavily in the past. In many cases it is also hard to make the connection between human disturbance and what is going on in the populations.

Section 10 ToR H (Marine mammals – COBAM indicators)

R1

This was the most complex ToR. The biggest issue I see here is the problem of dealing with targets that are put in such general terms that they cannot be quantified or measured. The WG has done a good job trying to put some operational definitions (e.g. IUCN reduction criteria) but it is sometimes hard to pick them out. A table with the general target and a more measurable one would be helpful. In cases where the proposed target cannot be realistically met, it could be highlighted. These should also be carried through to Annex 3 (e.g. as done for grey seal pup production).

All of the objectives are to recover (or maintain marine mammal) populations. I do not know if these were given beforehand but the reality is that scientists must follow the management objectives which they do not set and for seals, they may be quite different in some places. The proposal for a meeting between managers, scientists and stakeholders is critical although I caution that the suggestion that scientists provide model with some scenarios will only work if managers can provide some alternative objectives beforehand. In my experience, without some initial decisions by managers, the list of 'possibles' raised at these meeting become practically unlimited.

Using changes in pup production as a proxy of changes in population size (e.g. P27 and repeated elsewhere) can be inappropriate in some cases. For example, in NW Atlantic harp seals, annual fecundity rates vary greatly according to environmental conditions (e.g. Sjare and Stenson, 2010. Changes in the Reproductive Parameters of Female Harp Seals (*Pagophilus groenlandicus*) in the Northwest Atlantic. ICES J. Mar. Sci. 67:304–315). As a result, pup production can vary significantly while total population remains relatively constant (Hammill, M.O., G.B. Stenson, T. Doniol-Valcroze and A. Mosnier. 2012. Estimating carrying capacity and population trends of Northwest Atlantic harp seals, 1952–2012. Canadian Science Advisory Secretariat Res. Doc. 2012/148. iii + 31 p.). This may be more important among populations that are close to carrying capacity and/or live in highly variable environments but it is stated in a number of places that some of these populations may be near carrying capacity and so the use of pup production estimates can be difficult to interpret without information on fecundity.

Many of the figure and table captions are too brief to be of much use. For example, if Table 4 is proved by the EC's general evaluation matrix it should be stated. Otherwise it appears as if the WG may have developed the criteria. A clear separation between what was provide by other groups and developed by the WG throughout the report is needed. Three of the panels in Figure 3 show colour that is not explained. Also, there are two Figure 4s. The first one showing track lines can probably be removed. Figure 9 is difficult to interpret and should be redrawn.

For species such as harbour seals that do not occur throughout the entire area under consideration, it should be noted or a map of distribution provided. That would help explain, for example, why there is no discussion of assessment units in southern region of IV in Table 1.

10.2.8 (P13). The WG states that a lack of genetic differentiation is due to movement between the two areas. They need to be careful when making these statements (and they do say 'suggestion'). The lack of genetic differentiation can be the result of very small numbers of movements that, for management purposes, may not be important. They can also be because two populations separated relatively recently and differentiation has not occurred. Northwest Atlantic and Greenland Sea hooded seals are not genetically different but are treated by WGHARP as separate populations based upon know movements and a variety of other criteria. I am not suggesting that the recommendation is incorrect but just that the WG be aware of the issue.

10.2.2 It is stated that precision of estimates will increase as survey effort increases. Although this is usually the case, occasionally it does not happen (for example if distribution is very clumped). This should be acknowledged by stating that it is likely (or usually or in the majority of cases) rather than an absolute statement.

The WG states in several places that the use of stranded or bycaught animals provides good biological data. Examination of these animals is important and often our own source of information but it has to be recognized that they may not be a random sample. Interpreting contaminant levels in stranded St Lawrence beluga has been an issue of considerable debate and often samples of bycaught porpoise will be bias towards younger animals. Again, it will not affect the general advice but tempering some statement to acknowledge that the possibility of bias may exists in some circumstances would be prudent.

10.2.3 While stranding provides alternative information on distribution, bycatch will do the same. I believe it is mentioned elsewhere but it could be added here. Stenson *et* *al*. (2011. Using bycatch data to understand habitat use of small cetaceans: lessons from an experimental driftnet fishery. ICES J. Mar. Sci. 68:937–946.) provides an example.

Table 5 indicates that some data were obtained from a PhD thesis. It would be better to provide an indication of how the data were obtained or the type of data used.

Seals- Baseline (P39) Given the knowledge that many marine mammal populations are recovering, WGHARP accepts the 'maximum population estimated or observed' as the baseline for comparison to precautionary reference levels. In fact, the approach used by WGHARP (described in their reports) for a precautionary approach to the management of harp and hooded seals may be one that WGMME might wish to consider for other species. Canada is developing the approach so that it can be applied to other marine mammals (Stenson, G. B. M. Hammill, S. Ferguson, R. Stewart, and T. Doniol-Valcroze. 2012. Applying the Precautionary Approach to Marine Mammal Harvests in Canada. CSAS Res. Doc. 2012/107).

Grey seal abundance (P42–43). There are new data on abundance of grey seals in eastern Canada. The population on Sable island is no longer increasing at 12.8% (statement on P42) while the most recent model runs suggest that the gulf component is still increasing (Table 8). The data are given in the new Science advisory report that will be release next week on the DFO website (www.dfo-mpo.gc.ca/csas/publications). The reference is given below.

For both of the tables indicating current and known plans for monitoring of grey and harbour seals (P 55–59), it would be useful to have some indication of when the time-series began in order to understand what trend data may be available. Table P 55 what is meant by the phrase 'minimum required'?

10.5 Advice on Seals It is proposed that the aerial surveys for grey seal pup production be consistent between years. However, it is important to know if the population is expanding and to identify any new pupping areas that are developing. Otherwise, they will no longer provide a valid index of population. The same can apply to harbour seal haul-outs.

In the same section, it states that environmental covariates be considered. While this is true in general I am not sure how the examples (state of tide, time of day) apply to grey seal pup surveys. These are more appropriate to harbour seal haul-out surveys.

10.7 Recommendations (P78). The WG recommends that M1 and M2 be subsumed with in M3 and $ and therefore be removed from the list. Given that M5 (pup production) is a component of M3, should it also be subsumed? Pup production of harbour seals are difficult to obtain and not used very much while grey seal abundance is based upon a population model using pup production.

Is there any need to get acceptance of the 'measurable' criteria the WG proposes for the targets? If so, it should be included as a recommendation.

R 2

- M1. "Distributional range and pattern of grey and harbour seal breeding and haul-out sites, respectively" and M2. " Distributional range and pattern of cetaceans species regularly present"

Very little information is given regarding indicator M1 in the documents provided for the review (is there something missing?). However given that the indicators M1 and M2 are clearly linked to M3, M4 and to some extent also to M5 and given the problem in defining proper baselines, the suggestion from the EG to drop these indicators and include them within M3 and M4 is relevant. Information regarding distribution of breeding sites and haul-outs are available from the monitoring programs and could preferably be integrated into M3 and M4. Including information about distribution of breeding sites and haul-outs for seals and distribution of cetaceans is important since such changes are relatively easy to detect, even if they might be problematic to interpret, and should give an indication for further studies.

- M4. Abundance at the relevant temporal scale of cetacean species regularly present

Top predators like cetaceans and seals comprise an important part of biodiversity. Information regarding abundance and distribution is important to be able to assess GES according to MSFD. But the relevance of an indicator is also dependent on the possibilities to assess relevant data, and also to be able to draw conclusions regarding drivers causing the observed changes. Monitoring of cetaceans at sea is often costly and large-scale surveys are therefore occurring less frequently, making it difficult to validate the observed changes. It is important that sufficient resources is allocated for monitoring and that monitoring programs are well coordinated within the region and that other means of collecting relevant information for example through collection of stranded and bycaught animals are employed in order to investigate possible anthropogenic impact.

- M6. Numbers of individuals within species being bycaught in relation to population

The habitat directive requires a monitoring of bycatches and killing, and that such mortality should have no negative impact on the population. Bycatch is probably one of the most important factors causing human induced mortality in marine mammal populations. And bycatch limits is a key factor for achieving GES according to the MSFD. But reliable estimates of both bycatch rate and population size are often missing for marine mammal populations, as well as data for historical bycatch estimates. The target should be set so that conservation objectives could be met, but with the current situation with unreliable estimates of both bycatch rate and population size, and a lack of specific conservation objectives it is hard to make an advice. I therefore agree with the EG that it is an urgent need to define such objectives. Alternative approaches such as using current bycatch numbers as a baseline seems less productive, constant or even increasing bycatch in an increasing population could be a sign of a reduced bycatch problem as well as constant or even reduced bycatch in a decreasing population could be a sign of the opposite. A focus in future should be to find a consensus regarding conservation objectives and to define practical monitoring procedures for all fisheries as well as for aquaculture operations.

R3

With respect to Section 10, the WGMME has provided a very thorough review of all relevant information regarding OSPAR biodiversity indicators 1, 2 and 4, and has provided answers to all the various parts of the request. With respect to OSPAR biodiversity indicator 6, the WGMME report refers to Section 5. Although Section 5 deals with different ways of setting limits for bycatch, it does not deal with the other aspects of the request relevant to this indicator. However, most if not all of the information relevant to these other aspects can be found in the reports of WGBYC. I agree completely with the WGMME recommendation that the European Commission give serious consideration to ICES offer to host a workshop to review different mechanisms for setting safe bycatch limits.

Section 5 ToR C (Bycatch Limit algorithm)

R 1

5.3 The limitation of these methods should be made clear. Many of these methods assume that carrying capacity can be estimated and that it is relatively constant. Estimating K is not a trivial exercise and some discussion of this may be warranted. I am not sure how the RMP actually deals with K and so someone more familiar with it should be consulted to see if anything needs to be added.

The methods developed through simulation often assume that data-poor species can be assumed to be similar to those that are better known. This is the basis for the PBR and the default for R_{MAX}. However, modelling of the White Sea/Barents Sea harp seal population indicate that using the default R_{MAX} could result in a PBR estimate that would result in a population decline. This is described in the latest WGHARP report.

Section 9 Tor G (Interactions between wild and captive fish stocks)

R 1

Table 9.1 presents all of the information on aquaculture production that is available. It would be useful to have a summary of the total amount within the OSPAR marine area under consideration.

9.3 Interactions. It think a statement that,' in general, interactions between marine mammals and aquaculture are not well studied in most areas' might be warranted. The list of mammals associated with aquaculture appears correct for the NE Atlantic but if it is the entire North Atlantic, *Lutra canadensis* should be added as it is an issue in NW Atlantic waters. I hesitate to say that 'in the majority of instances the associations … are benign". In my area, this would not be the case. Perhaps change this to 'Often' rather than the majority.

9.3.1 Damage. It would be useful if they could separate harbour from grey seals. In my experience, grey seals tend to cause more damage, in part because of their larger size. Their behaviour also tend to differ.

9.4.4 Deliberate killing. Are there any estimates on the amount of 'non-reporting' that may be occurring? This can be quite high in some places.

The numbers of grey seals taken under nuisance seal licences in eastern Canada is much higher than reported in the Pacific. The most recent estimates are given in the new science advice for grey seals that will be published on the web next week. It also has the information of grey seal abundance mentioned above.

Table 1. Reported removals from the NW Atlantic grey seal population over the last years. [1] the nuisance seal estimate is based on the number of seals reported removed, divided by the reporting rate.

	2008	2009	2010	2011	2012	2013
Commercial harvest 1+	1,471	263	58	215	218	106
Science collections	0	0	0	320	90	0
Nuisance seals[1]	3,018	5,218	1,853	1,722	5,428	3,525

The proper reference is: DFO. 2014. Stock Assessment of Canadian Grey Seals (*Halichoerus Grypus*). DFO Can. Sci. Advis. Section Sci. Advis. Rep. 2014/010.

R 2

The expert group defined the request from ACOM as "review the effects of aquaculture on marine mammals, and where possible provide examples of management solutions." The group makes a good summary of what is known regarding interactions between marine mammals and aquaculture. Marine mammals, especially seals, foraging in or close to fish farms might cause some of the interactions between wild and captured fish, by damaging pens. More effort should be made in quantifying such damage and also in methods to reduce such damage.

R 3

With respect to Section 9, the WGMME has provided a thorough review of the available information on the interactions between aquaculture and marine mammals and provided examples of management solutions that have mitigated effects of marine mammals on aquaculture. However, the last part of the request, i.e. to *"- -recommend which pressures have sufficient documentation regarding their impacts to implement relevant monitoring and suggest ways forward to manage these pressures."* has not been covered adequately by the WGMME. This is probably because there is at present insufficient information regarding the pressures exerted by aquaculture on marine mammals, as most research has focused on the effects of marine mammals on aquaculture where the major problems seems to be.

Annex 5: WGMME terms of reference for the next meeting

The **Working Group on Marine Mammal Ecology** (WGMME), chaired by Begoña Santos (Spain) and Graham Pierce (UK), will meet in London, 16–19 February 2015:

1) Review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals; specifically. This will contribute to the work required for the MoU between the European Commission and ICES to "*provide new information regarding the impact of fisheries on other components of the ecosystem including small cetaceans and other marine mammals*…" and to aid "*scientific and technical developments in the support of the Marine Strategy Framework Directive, such as by designing marine monitoring and assessment programmes, identifying research needs and methodologies advice*". OSPAR is also seeking advice from ICES in relation to the development of indicators and targets for determining Good Environmental Status (GES) under MSFD to which this will contribute;

2) Review North Atlantic information on negative and positive ecological interactions between grey seal (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) populations;

3) Update on the development of database for seals, determining its contribution and the state of operationalisation for assessments of seal GES under MSFD;

4) Review and evaluate multispecies models that incorporate marine mammal consumption to assess marine mammal impacts on fishery resources, and make recommendations for improvements in input data and assumptions for the North Atlantic.

WGMME will report for to the attention of the Advisory Committee.

Annex 6: Recommendations

Recommendation I

WGMME strongly supports the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2016 and **recommends** that it is supported by all range states and by ICES, ASCOBANS and the European Commission. Continuation of these surveys is essential to the accurate estimation of absolute abundance for several species that are required for reporting under the Habitats Directive and the Marine Strategy Framework Directive.

Recommendation II

As part of the reforms to the Common Fisheries Policy and the Data Collection Framework, the European Commission requested that ICES provide advice on the use of management frameworks and other mechanisms for determining safe bycatch limits in 2013. The ICES response noted that further work in this area would be required and that: '*This could be in the form of a workshop for invited participants representing managers, scientists and stakeholders. As stressed in the advice, input from management and from the "societal" side is crucial for such a process. We would envisage attendees from relevant parts of the European Commission (at least DG Mare and DG Environment), Member State fisheries authorities, the RACs, relevant intergovernmental bodies (Regional Seas Commissions, ASCOBANS and ACCOBAMS) and relevant NGOs.* Given that the lack of agreed conservation objectives is the primary reason stopping further development, WGMME **recommends** that European Commission give serious consideration to ICESs offer to host a workshop, with the objective of reviewing different mechanisms for determining safe bycatch limits and finalising conservation objectives for a bycatch limit approach that would enable conservation aspiration to be met.

Recommendation III

WGMME **strongly recommends** that ICES members of the OSPAR region provide data so that the seal database can be maintained and updated regularly. Such development is considered essential to future MSFD assessments of the OSPAR core set of indicators for seals.

Recommendation IV

There is a wide range of monitoring methodologies available to assess marine mammals at marine renewable energy development sites, but not all techniques are equally appropriate to all sites. Moreover, assessing the suitability of techniques and the quality of resulting survey data can be hampered by incomplete reporting of methodological details by developers. Commercial sensitivities may further complicate efforts by regulators and others to compare monitoring techniques on their respective merits.

WGMME **recommends** that regulators and policymakers should require the use of open, transparent and reproducible survey and monitoring methodologies to assess potential impacts on marine mammals. Furthermore, for line-transect surveys, the data should be fit to provide absolute densities. For all monitoring, the use of established and peer-reviewed methods is encouraged, acknowledging that new innovations or methods may arise. Methods associated with such new techniques should be sufficiently well described so that conclusions arising from these techniques are reproducible. Data from surveys should be made publicly available in formats that allow future reanalysis (for example using JCP-type protocols).

Recommendation V

WGMME **recommends** that the OSPAR consider the proposals made in relation to their request. In summary these are:

1) Adoption of the proposed assessment units for grey and harbour seals in the OSPAR Maritime area; assessment units for the more common cetacean species are also proposed.

2) Focus monitoring and assessment on M3 and M4 (trends in abundance), with M1 and M2 (range and pattern of distribution) being removed from the list of common indicators; and subsumed within M3 and M4, respectively. It is not possible to propose a firm and measurable baseline, metric and target for common indicators M1 or M2. Distribution changes should act as warning signals and research should be carried out to investigate the causes of those changes, especially to determine if they have an anthropogenic cause.

3) Consider the technical and scientific advice provided on options for setting targets, determining baselines and associated monitoring requirements. Some standards for monitoring are suggested.

4) With respect to M6 (bycatch), it is not possible to progress this indicator significantly. Further work and collaboration between the European commission, ICES and OSPAR is required. WGMME have recommended that the European Commission give serious consideration to ICESs offer to host a workshop, with the objective of reviewing different mechanisms for determining safe bycatch limits and finalising conservation objectives for a bycatch limit approach that would enable conservation aspiration to be met.

5) OSPAR ICG-COBAM and HASEC should continue evaluating the proposed common PCB indicator for use within Descriptors 1 and 8, respectively.

The WGMME recommends that collaboration with ICG-COBAM is maintained for the continued development of the mammal common indicators.