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

5–8 March 2012

Copenhagen, Denmark

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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

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Executive summary

The Working Group on Marine Mammal Ecology (WGMME) met at ICES in Copenhagen, Denmark from 5 March to 8 March 2012. Eunice Pinn chaired the meeting of 21 participants, representing eight countries.

Three ToRs were addressed, the first covered new information on abundance and provided advice on suitable management units while the second looked at potential marine mammal indicators building on the work undertaken last year and also that of OSPAR ICG-COBAM expert group on marine mammals. The third ToR reviewed the development and potential effects of wave energy converters on marine mammals and provided recommendations on future research needs. One ToR, on the development of the seal database, was deferred to 2013.

The WG built on the work of the ASCOBANS/HELCOM small cetacean population structure workshop to determine Management Units (MUs) for the more common species as such information is relevant to the development of biodiversity indicators. Based on the available information, there were single MUs in European North Atlantic for common dolphin (*Delphinus delphis*), white beaked dolphin (*Lagenorhynchus albirostris*), white sided dolphin (*Lagenorhynchus acutus*) and minke whale (*Balaenoptera acutorostrata*). For bottlenose dolphin (*Tursiops truncatus*) there are ten separate units closely associated with the mainly resident inshore populations in the European North Atlantic and a separate MU for the wider ranging mainly offshore animals. For harbour porpoise (*Phocoena phocoena*), MUs are proposed for the Iberian Pennisula, Bay of Biscay, Celtic Sea (including SW Ireland, Irish Sea and Western Channel) and NW Ireland/West Scotland and the North Sea. The MUs for harbour porpoises will need to be revisited as indicators for MSFD become better defined. It is likely that MUs will need to be aligned with ICES rectangles to enable the calculation of accurate bycatch estimates. For the purposes of MSFD, it maybe that consideration of the species will need occur at the regional seas level (e.g. North Sea).

OSPAR's ICG-COBAM, as part of the development of the Advice Manual on MSFD indicators, developed a series of summary sheets for the 'common' indicators. Prior to publication, the sheets for marine mammals were made available to WGMME for further consideration. Biodiversity indicators covering seal distributional pattern and abundance were assessed against available information and suggestions made for the most appropriate target and metrics. These were closer to OSPAR's EcoQOs than the more generic approach in the ICG-COBAM sheets. A similar consideration was also given to suggested indicators for cetacean distributional pattern and abundance. It was recognised that for cetaceans in particular, a transboundary approach is essential to both the monitoring and the assessments. Without such an approach, the value of the information collected and the accuracy of the status and/or indicator assessments made will be much lower. Further development of these distribution and abundance biodiversity indicators will be undertaken next year.

As part of the further development of indicators, bycatch was the only indicator suggested that had a clear link with a particular human activity. The indicator metric proposed by ICG-COBAM was very clearly linked to OSPAR's EcoQO on harbour porpoise bycatch in the North Sea. Bycatch is been considered on a regular basis in previous years by both WGMME and the Working Group on the Bycatch of Protected Species (WGBYC). With pressure for the rapid development of biodiversity indicators for good environmental status through the MSFD, it is essential that they are based on sound science and take a pragmatic approach to the incorporation of fisheries

data. As such, it was proposed that a management framework approach is adopted (rather than the EcoQO approach) and further developed in 2013 for relevant species.

The marine renewable industry is a rapidly developing sector. In past meetings, WGMME looked at the effects of windfarms (2010) and tidal devices (2011) on marine mammals. In 2012 it was the turn of wave energy converters. Wave energy converters (WECs) are at a relatively early stage of development when compared to other renewable energy technologies. This is reflected in the lack of knowledge of effects that these devices might have on the marine environment in general and, therefore, a lack of information available for environmental consenting. In order to satisfy national and international requirements (e.g. the Habitats Directive), monitoring schemes need to gather baseline information before construction begins, as well as continued impact monitoring during the construction, operation and decommissioning phases of the deployment. Broadly, monitoring must take place at spatial and temporal scales that are appropriate to assess impacts upon marine mammals at the population level, although this rarely happens. It is, therefore, essential that full advantage is taken of test deployments and early arrays to gather information on the actual interactions between devices and wildlife. A review of such work will be undertaken in 2013.

1 Introduction

The Working Group on Marine Mammal Ecology (WGMME) met in Copenhagen from 5 March to 8 March 2012. The list of participants and contact details are given in Annex 1.

The Working Group gratefully acknowledges the support given by several additional experts that kindly provided information and/or reports for use by WGMME and reviewed parts of the report. The WG acknowledges the support given to us by Marisa Ferreira, Jack Lawson, George Lees, Daniel Moysey, Ian Davies, Emer Rogan, Michelle Cronin, Silvia Casini, Anita Gilles, Meike Scheidat, Mike Lonergan, Sonia Mendes, Callan Duck, Debbie Russell, and Alexandros Karamanlidis who all provided information and reports that were used at the meeting, contributed to the report and/or contributed to the CRR on surveillance and monitoring of marine mammals in the ICES area.

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 Adoption of the agenda

The following Terms of Reference and the work schedule were adopted on March 5th 2011.

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

b) Develop biodiversity indicators in support of policy drivers, and develop indicators that are robust to expected uncertainties in data and/or to provide a quantitative analysis of the potential effects of data limitations on indicator performance;

c) Outline and review the effects of wave energy devices on marine mammals and provide recommendations on research needs, monitoring and mitigation schemes;

d) Update on development of database for seals, status of intersessional work.

In addition, it was agreed to finalize production of the Cooperative Research Report on the framework for surveillance and monitoring of marine mammals applicable to the ICES area;

WGMME will report to the attention of the Advisory Committee (ACOM) by 23 March 2012.

Supporting information

Priority:	High, as only group that can support requirements in ToR a.
Scientific justification and relation to action plan:	a) This work is required under MoU between the European Commission and ICES: "provide new information regarding the impact of fisheries on other components of the ecosystem including small cetaceans and other marine mammals…" b) Fulfills a recommendation for action from WKMARBIO c) This is completion of the revew of the effects of renewable energy on marine mammals within the ICES Area. It addresses the research topic "Influence of development of renewable energy resources (e.g. wind, hydropower, tidal and waves) on marine habitat and biota" within the ICES Science Plan d) This will facilitate future work of the WG
Resource requirements:	No specific requirements beyond the needs of members to prepare for, and participate in, the meeting.
Participants:	The Group is normally attended by some 20–25 members.
Secretariat facilities:	None.
Financial:	No financial implications.
Linkages to advisory committees:	WGMME reports to ACOM
Linkages to other committees or groups:	SCICOM SSGSUE
Linkages to other organizations:	

3 ToR a) Population structure

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

3.1 Management frameworks and management units

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). Harbour porpoise populations were modelled under various scenarios of bycatch and target population size using a very simple deterministic population dynamics model with assumed maximum rate of increase of 4%. It applies to a "biological" population with independent population dynamics. The 1.7% figure is the rate of total removals from a population that would still allow the harbour porpoise population to achieve 80% of its carrying capacity over a very long time horizon (a proxy for a sustainable population). The figure has subsequently been adopted by ASCOBANS as the rate above which bycatch would become "unacceptable"; noted by a North Sea Ministerial meeting, developed into an EcoQO by OSPAR and accepted by the European Commission (Anon., 2010) as the level above which ICES might advise that mitigation measures would become necessary. The OSPAR EcoQO on porpoise bycatch in the North Sea states that "*Annual bycatch levels should be reduced to be below 1.7% of the best population estimate*".

Notwithstanding that the figure of 1.7% is a gross over-simplification (as described in IWC 2000), if this management target is to be applied to management units (MUs) for harbour porpoise, the animals living in the areas defined by these MUs 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.

WGMME reiterates the need for management targets to be determined in the context of explicit conservation and management objectives. It draws attention to the conclusions of its discussions on the use of the management procedure approach to determine safe limits to small cetacean bycatch in the context of specified conservation objectives (WGMME 2008, Section 4 and WGMME 2009, Section 7.2), including the strong recommendation that the bycatch management procedures developed under the SCANS-II and CODA projects (SCANS-II 2008; Winship 2009; CODA 2009) should be taken into consideration by DG MARE when reviewing EU Regulation 812/2004 (WGMME 2009, Section 7.3). WGMME (2010) recommended that ICES be encouraged to "move away from implicit and automated conservation targets and towards the explicit definition and justification of target population sizes and management objectives". Subsequently, ICES advice to the European Commission in 2010 (Section 1.5.1.2) included the following recommendations:

'(i) ICES could provide advice to the Commission on the various approaches to establishing specific conservation and management objectives to manage the impacts of fisheries on marine mammal and seabird populations.

(iii) 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.'

One of the targets of the ASCOBANS' North Sea Conservation Plan for Harbour Porpoises is '*to finalise a population dynamics modelling framework for evaluating the effect of bycatches (and other anthropogenic activities) on harbour porpoises in the North Sea that anthropogenic activities do not prevent agreed conservation goals being met.... building upon the advances made by the IWC/ASCOBANS working group, the ICES/SGBYC and the SCANS II project and the recommendations therein and other Actions (2, 3, 4, 7) of this plan including: agreement of operational management objectives by policymakers; finalisation and scientific implementation of a management procedure by scientists; agreement by policymakers to develop and implement management advice based on the results of the management procedure' (ASCOBANS, 2009).* Similarly, the ASCOBANS Working group on Bycatch has also indicated that '*the IWC/ASCOBANS working group had recommended a management procedure approach using simulation studies to develop algorithms for setting limits to achieve management objectives*' (ASCOBANS, 2012).

These and other issues relating to management rules for marine mammal exploitation have recently been discussed in the literature (Lonergan 2011; Cooke *et al.*, 2012; Lonergan, 2012). The processes of management strategy evaluation and the management procedure approach to fisheries management, pioneered by the IWC in the development of the Revised Management Procedure (RMP), which aims to maintain cetacean populations above a fixed proportion (72%) of their carrying capacity, and used in the development of the bycatch management procedures described above, are the topic of a session at the World Fisheries Congress in Edinburgh in May 2012. The only appropriate way to incorporate the concept of management units that are not biological populations with more or less independent dynamics is within management frameworks such as those developed for setting limits to safe bycatch for harbour porpoise and common dolphin (SCANS-II 2008; Winship, 2009; CODA 2009). These projects considered two procedures, one based on PBR which used a single, current estimate of absolute population size as input and one based on the RMP (termed the Catch Limit Algorithm or CLA) that used a time-series of estimates of absolute population size and estimates of absolute bycatch as input. Both procedures were tuned to three different potential conservation objectives:

i) Median population at 80% of carrying capacity after 200 years;

ii) A 95% probability that the population would be at or above 80% of carrying capacity after 200 years;

iii) Worst case scenario with biased input data and a 95% probability that the population would be at or above 80% of carrying capacity after 200 years.

Based on analysis of data on harbour porpoise in the North Sea, Winship *et al*. (2006) suggested that despite being more complex, the advantages conferred by the CLA procedure were sufficient for it to be considered as the best option. They also recommended that management objectives should be precisely specified and that the judgement of which tuning to use could be based on an assessment of the available information.

To enable the development of bycatch indicators, WGMME reiterates its **strong recommendation** that the bycatch management procedures developed under the SCANS-II and CODA projects (SCANS-II 2008; Winship, 2009; CODA 2009) should be taken forward to develop management frameworks for marine mammal bycatch at a European level. The development of bycatch indicators for the MSFD should be based on such an approach rather than a direct transfer of the simplistic percentage approach. It is recommended that WGMME and WGBYC collaborate to progress this work during 2013 for harbour porpoises, common dolphins, as well as grey and harbour seals, as part of the MSFD bycatch indicator developments.

3.1.1 Defining management units in practice

The report of the ASCOBANS/HELCOM small cetacean population structure workshop included a useful summary of how information on population structure can be used to determine Management Units (MUs) and the multiple difficulties inherent in trying to achieve this (Evans and Teilmann, 2009). In specifying how the available information had been used to propose MUs for the small cetacean species considered, the workshop stated:

"*In general, the integration of both genetic and ecological markers is necessary to obtain the best possible indication of relevant stock structure. A major challenge that still needs fully addressing is how to integrate these rather different lines of evidence, and what time frame is most appropriate to consider here in the context of conservation management. For the time being, we consider a few generations (equivalent to low tens of years) as the appropriate time frame for defining a management unit, and we identify an MU as a group of individuals for which there are different lines of complementary evidence suggesting reduced exchange (migration/dispersal) rates. Ideally, one should set quantitative parameters (e.g. maximum of ten percent migration per generation), but in most cases we do not have the information as yet to do this, nor has the theoretical framework for integration of different evidence bases been fully developed*."

WGMME **agrees** that this is a reasonable approach to take at the present time.

At this meeting, WGMME reviewed the information available on management units available for the most commonly encountered species in the eastern North Atlantic. The report of the joint ASCOBANS-HELCOM small cetacean population structure workshop (Evans and Teilmann, 2009) had been the basis for WGMME discussions on management units (hereafter referred to as MUs) for common dolphin, harbour porpoise and bottlenose dolphin in 2009, 2010 and 2011, respectively. Evans and Teilmann (2009) contains similar information on white-beaked and white-sided dolphin.

3.1.1.1 Harbour porpoise

In 2010, WGMME endorsed the MUs for harbour porpoise proposed by the ASCOBANS-HELCOM small cetacean population structure workshop, shown in Figure 3.1. This year WGMME reviewed these MUs and its recommendations from 2010, which were:

1) WGMME strongly recommends that the Iberian harbour porpoise population should be given high priority for conservation, as a consequence of its presumed small population size, low genetic diversity and likely susceptibility to habitat degradation.

2) WGMME strongly recommends immediate action by the Spanish and Portuguese governments in monitoring and conserving the Iberian harbour porpoise population.

3) WGMME recommends to ASCOBANS the establishment of a separate conservation plan for the harbour porpoise Inner Danish Waters MU.

4) WGMME recommends to ASCOBANS to take into account the existence of the two newly designated harbour porpoise Management Units in the North Sea, northeastern North Sea and Skagerrak and southwestern North

Sea and Eastern Channel, within their harbour porpoise North Sea conservation plan; with the inclusion of the Shetland Islands, Skagerrak and northern Kattegat within the northeastern North Sea MU.

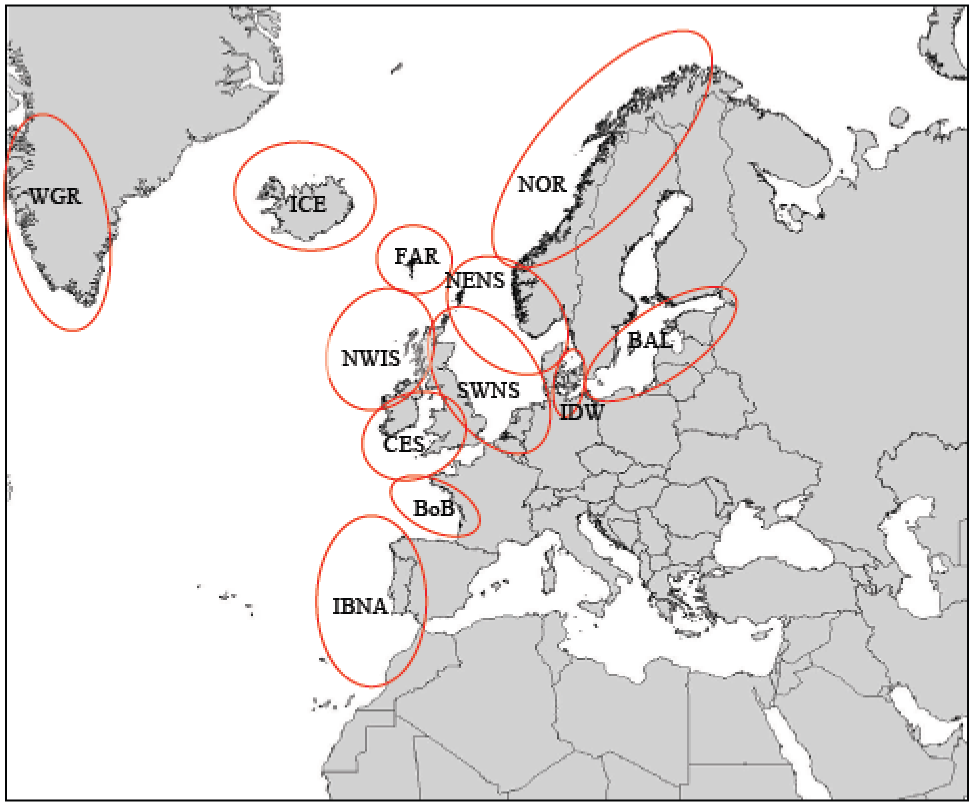

Figure 3.1. ASCOBANS proposed harbour porpoise management units (taken from Evans and Teilmann, 2009). NOR – Northwest/Central Norway and Barents Sea, NENS – Northeastern North Sea and Skagerrak, SWNS – Southwestern North Sea and Eastern Channel, IDW – Inner Danish Waters, BAL – Baltic Sea, CES – Celtic Sea (plus SW Ireland, Irish Sea and western Channel), NWIS – Northwest Ireland and West Scotland, BoB – Bay of Biscay (west France), IBNA – Iberian Penninsula (NW Spain, Protugal and NW Africa).

Concerning recommendations 1) and 2), a programme was established in Portugal in 2010 to monitor the distribution and abundance of marine mammals (harbour porpoise, common dolphin, bottlenose dolphin and minke whale). This monitoring scheme is supported by the SafeSea (EEAGrants) and MarPro (LIFE+) projects and will continue systematic observations until 2015. The monitoring scheme is an integrated programme using several different approaches: aerial surveys, dedicated offshore surveys using SCANS/CODA methodologies, land-based counts and opportunistic boat surveys. A final report on this analysis will be produced by the end of 2012.

In Spain, work on monitoring harbour porpoise has been limited to the north of the country, through a programme developed by CEMMA for the Region of Galicia. The government of Spain has enabled the development of conservation plans for species on its protected species list; CEMMA with support from Fundación Biodiversidad has developed a conservation plan for the harbour porpoise. The future of this initiative is, however, uncertain following the closure of the relevant Government Department.

Concerning recommendation 3), a draft conservation plan for harbour porpoise in the western Baltic, the Belt Sea and the Kattegat has been developed for ASCOBANS. This will be reviewed and amended, as appropriate, at the 19th Advisory Committee meeting in March 2012 with a view to it being adopted at the 7th Meeting of the Parties being held in October 2012.

Concerning recommendation 4) and the splitting of the North Sea into two Management Units, WGMME noted: (a) the very strong difference in distribution of harbour porpoises in the North Sea observed in SCANS-II 2005 compared to SCANS in 1994; (b) the near continuous distribution of SCANS-II sightings across the southern and central North Sea and up the east coast of the UK (Figure 3.2); and (c) the widespread movements of animals radio-tagged off northern Jutland across the central and northern North Sea (Figure 3.3). Splitting the North Sea into two Management Units is therefore not supported by the data.

There was also some discussion regarding the possible division of the Celtic Sea (plus SW Ireland, Irish Sea and western Channel) MU into Celtic and irish Sea and some concern of the separation of the northwest Ireland and West Scotland MU from the Celtic Sea.

WGMME **recommends** a single North Sea Management Unit comprising ICES Area IV, and most of Division IIIa (Skagerrak and northern Kattegat) south to the most appropriate boundary with the Belt Seas MU. It should also include Division VIId; very few harbour porpoise are seen in the eastern Channel. The northern boundary with Division VIa is arbitrary. The MUs for harbour porpoises will need to be revisited as indicators for MSFD become better defined. When they are being developed, they will need to be aligned with the appropriate ICES rectangles to enable the calculation of more accurate bycatch estimates.

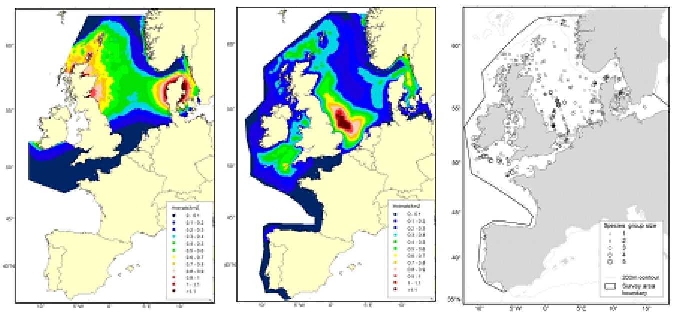

Figure 3.2. Modelled distribution of harbour porpoises in the European Atlantic from SCANS surveys in 1994 and SCANS-II surveys in 2005 and distribution of harbour porpoise sightings from SCANS-II (from Hammond *et al.*, in prep).

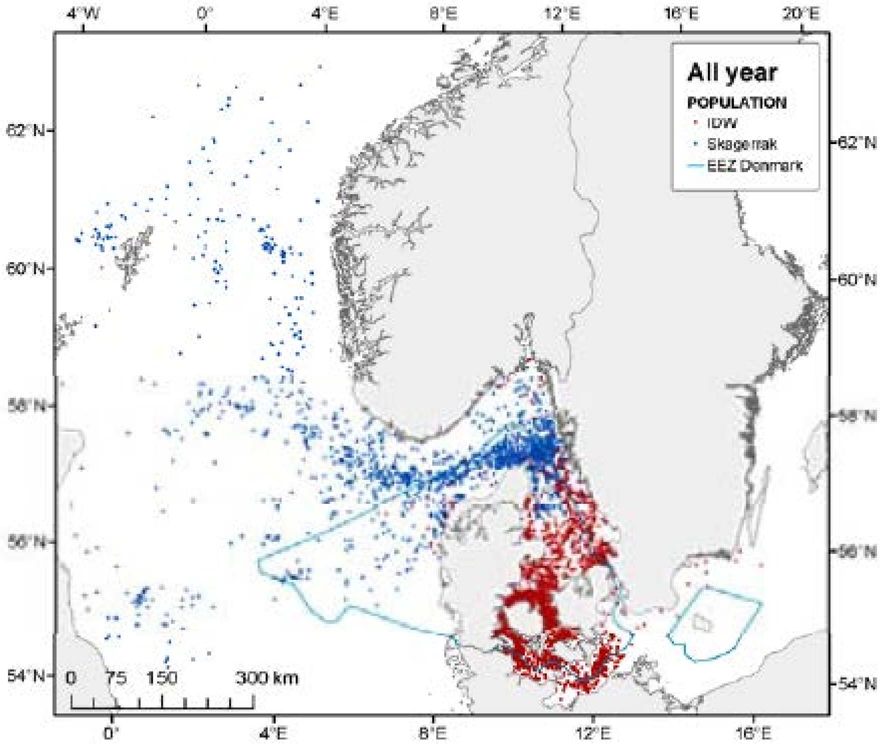

Figure 3.3. Locations (one per day) of 63 radio tagged porpoises. Porpoises tagged in the IDW are red and those tagged at the tip of Jutland are blue. (N = 63, n = 4287 locations). (Taken from Evans and Teilmann, 2009).

3.1.1.2 Common dolphin

The ASCOBANS-HELCOM workshop concluded that there was little evidence of genetic structure in the Northeast Atlantic suggesting that there was a single population, ranging from waters off Scotland to Portugal, but with separate populations in the Northwest Atlantic, and Mediterranean Sea (Figure 3.4; Evans and Teilmann, 2009). It further proposed that, due to the low genetic differentiation in this species, common dolphins in the Northeast Atlantic should be managed using an ecological stocks approach. However, although stable isotope and contaminant analyses suggest there may be some structuring of common dolphin populations within this region (see Caurant *et al.*, 2009), with a possible existence of neritic and oceanic ecological stocks, at present there are insufficient data to verify this or to designate separate "ecological" management units.

Following this, WGMME (2009) made the following conclusions and recommendations:

- Only one *D. delphis* population exists in the Northeast Atlantic ranging from waters off Scotland to Portugal, and separate populations have been reported in the Northwest Atlantic and the Mediterranean Sea; suggesting one management unit in the NE Atlantic, based on genetic data.
- All samples analysed for genetic analysis in the NE Atlantic were obtained from continental shelf and slope waters, and the oceanic waters of the Bay of Biscay, and therefore the management unit/area for *D. delphis* in the NE Atlantic is confined to this region.
- The actual distributional range of the population is not known. In order to assess what the distributional range of the population is, samples need to

be obtained from offshore common dolphins, and analysed using both genetic and ecological markers. This can only be undertaken by obtaining samples of skin and blubber from biopsies.

- The high haplotype diversity of control region suggests a large effective population size of common dolphins living in the NE Atlantic.
- As a consequence of the low genetic differentiation in this species on a whole, it is proposed that common dolphins in the NE Atlantic should be managed using an ecological timescale, i.e. managing ecological stocks. However, as a consequence of small sample sizes, data obtained to date using ecological markers are not adequate for describing the existence of ecological stocks in the Northeast Atlantic.
- Therefore, directed studies should be undertaken on assessing the existence of ecological stocks in this region using a large number of samples, obtained from all age/sex classes, and sampling animals over a large geographical area. A number of ecological markers such as heavy metals, stable isotopes, fatty acids, etc should be used.
- A genetically divergent lineage within the genus *Delphinus* has been identified in the NE Atlantic. This raises questions regarding to the taxonomic status of common dolphins in this region. Further analysis needs to be undertaken prior to establishing and implementing the existence of a separate evolutionary stock/species into a Northeast Atlantic common dolphin management plan.
- As a consequence of a lack of sampling of offshore common dolphins for genetic (and ecological) analysis, the WGMME recommends that the management unit/area for the *D. delphis* population in the NE Atlantic be confined to the continental shelf and slope waters, and the oceanic waters of the Bay of Biscay.
- The WGMME highly recommends that all the samples and data available (genetic and ecological) in the ICES area are analysed together, in order to get the most comprehensive picture of the population structure. Special emphasis has to be put on the understanding of the process underlying such a low genetic structure. In this respect, a paramount aspect is to determine whether the population structure consists in continuous gradation, with local habitat related variation, or if clearly delimited stocks can be identified. This requires a large scale study, incorporating samples from offshore waters, conducted at the level of individual, and using spatially explicit analysis.

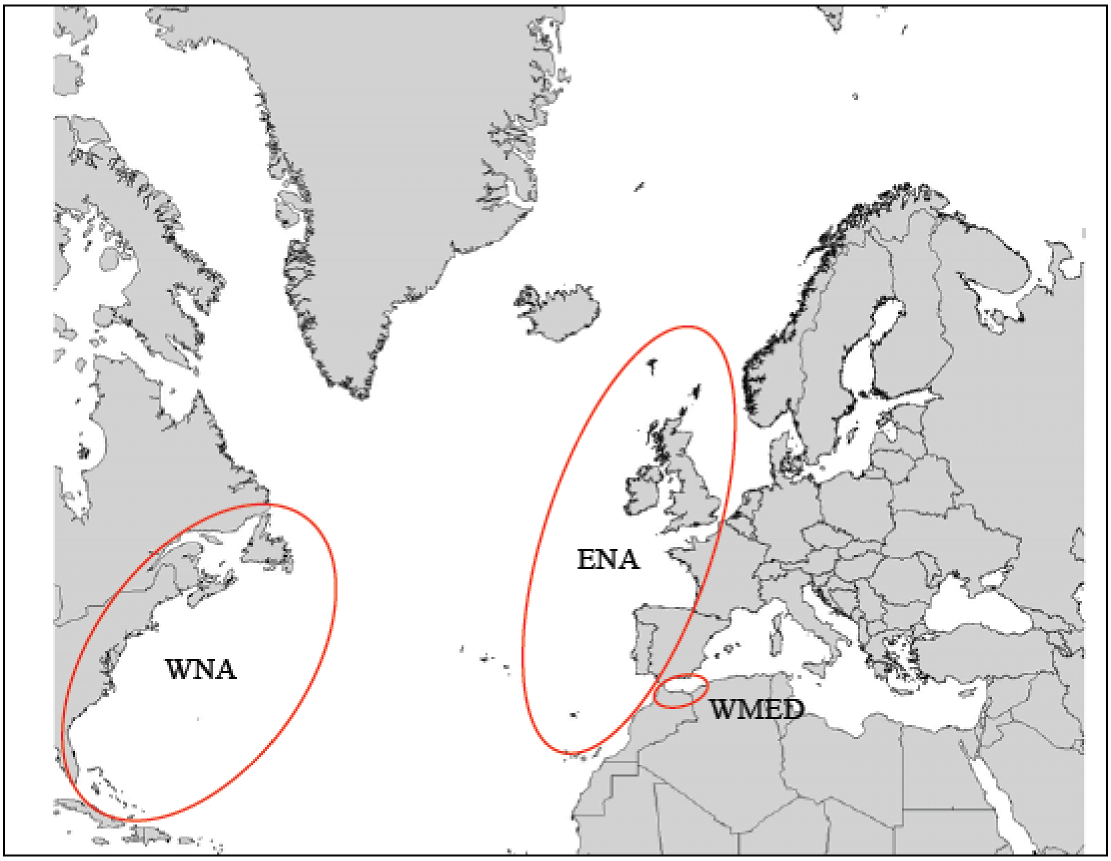

Figure 3.4. ASCOBANS recommended management units for common dolphin (taken from Evans and Teilmann, 2009). WNA – Western North Atlantic, ENA – Eastern North Atlantic, WMED – western Mediterranean Sea.

Based on the available information, WGMME **endorses** these previous conclusions and recommendations. Thus the European North Atlantic common dolphins are considered to represent a single management unit.

3.1.1.3 Bottlenose dolphin

The ASCOBANS-HELCOM small cetacean population structure workshop had proposed provisional Management Units (Figure 3.5).

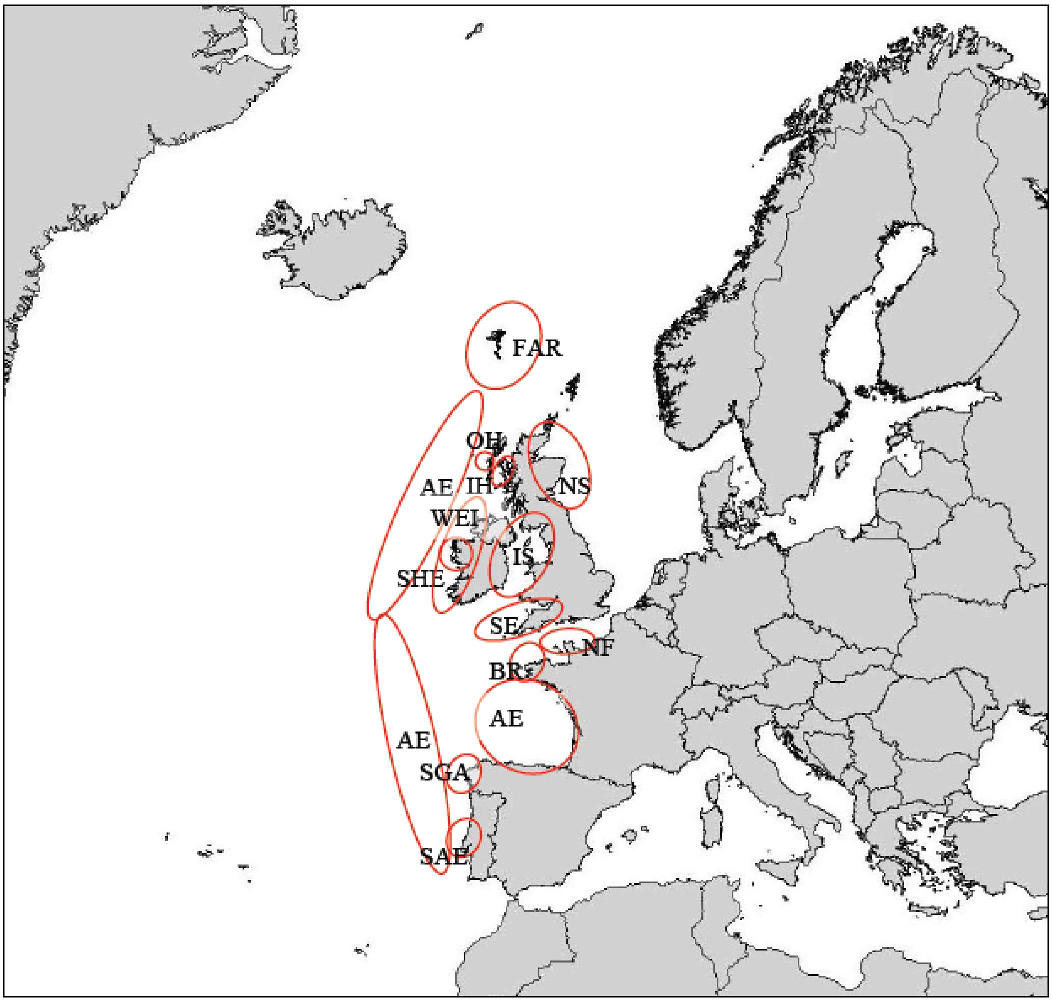

Figure 3.5. ASCOBANS recommended management units for bottlenose dolphin (taken from Evans and Teilmann, 2009). AE – Atlantic Europe, NS – North Sea, OH – Outer Habrides, IH – Inner Hebrides, IS – Irish Sea, SE Southern England, NF – Northern France/Channel Islands, SHE – Shannon Estuary (Ireland), WEI – Western Ireland, BR – Britany, SGA – South Galicia, SAE – Sado Estuary (Portugal).

WGMME (2011) concluded that the following work was needed:

- Further assessment of population structure in offshore waters/Atlantic Europe to discriminate population structure in this region.
- Further discrimination of population structure within coastal waters.
- Undertake photo-id studies of coastal populations in the southern distribution of its range in the Northeast Atlantic for establishing range movements, i.e. southern French and Iberian waters.
- Undertake other complementary approaches to assess population and ecological stock structure including skull morphometric analysis, assessment of parasite and contaminant loads, and variation in life-history parameters, and stable isotope analysis.
- As the existence of the Connemara-Mayo putative population was only identified very recently (Mirimin *et al.*, 2011), this highlights the importance of monitoring coastal areas in order to allow for the identification of such aggregations that may be locally adapted to specific areas.

- It is recommended that samples sizes for genetic analysis are increased from Iberia, Wales, western Ireland and Scotland and, where appropriate, biopsy samples are obtained from bottlenose dolphins.

The resident population of bottlenose dolphins in the Sado Estuary, Portugal, have been declining over the last three decades. Agusto *et al*. (2011) undertook a complete photographic census of the population, which produced a count of 24 animals; 19 adults, three subadults and two calves. The population is thought to be phylopatric and essentially closed, but given the likely importance that exchanges with neighbouring coastal groups may play, even if rare, the most adequate term to define this dolphin should be community and not population. They concluded that the social structure of the community is influenced by a combination of demographic characteristics and a stable and productive environment, which has led to a decrease in competition between individuals.

In Cornwall (SW Btitain), Pikesley *et al*. (2011) noted significant decreases in bottlenose dolphin sightings and pod size between 1991 and 2008 from incidental sightings/strandings. Bottlenose dolphin showed twin peaks in sightings in late spring and autumn and were found in shallow near-shore coastal waters particularly concentrated in harbour and bay areas often within or close to estuary mouths as well as around Land's End headlands (Figure 3.6). Field studies and photo identification suggested an emigration or loss of individuals from the area between 1994 and 1996.

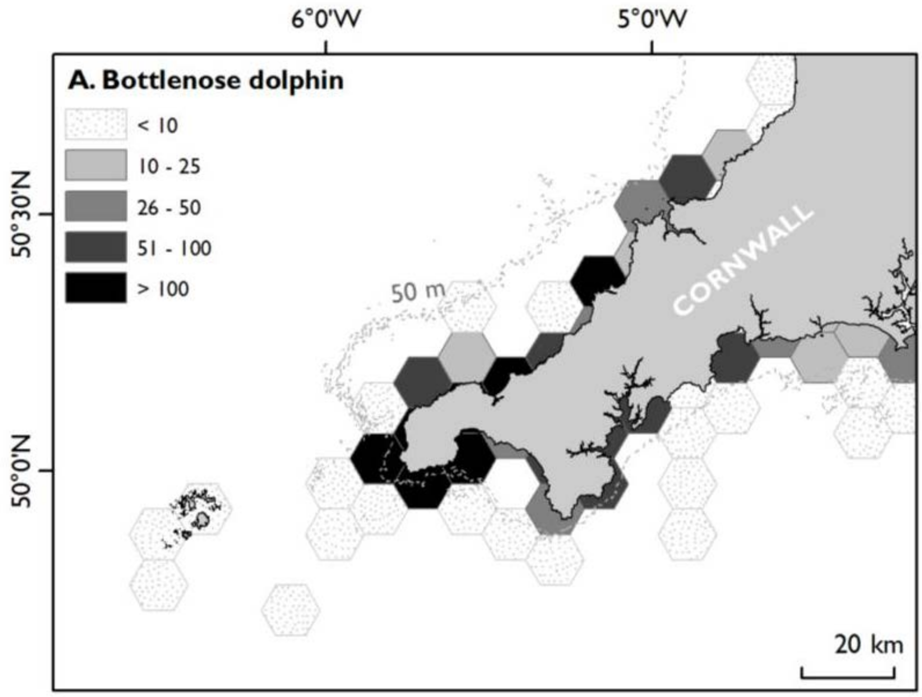

Figure 3.6. Hexagonal polygon binning density estimates for bottlenose dolphin sightings (n=2851) between 1991 to 2008. Taken from Pikesley *et al*. (2011).

Cheney *et al*. (2012) proposed the presence of three parapatric communities of bottlenose dolphins in Scottish coastal waters, each of a different size and with marked contrasts in their ranging patterns. On the west coast, there are two small and socially segregated communities of dolphins, one of which includes approximately 15 individuals that have only been recorded in the waters around the Sound of Barra, whereas the other is double that size and ranges more widely throughout the Inner Hebrides and mainland coasts. On the east coast, there is a population of nearly 200 interacting dolphins between the Moray Firth and Fife, with individual differences in ranging behaviour and site fidelity. Analyses of photo-identification data from multiple studies have also shown that bottlenose dolphins can make long-distance movements between the east and west coasts of Scotland, and further exchange between Scottish and Irish waters has recently been revealed (Robinson *et al*., in press). Whether these movements represent exchange between different coastal communities or interaction with more widely ranging offshore animals remains uncertain.

Based on the available information, WGMME **endorses** the MUs for bottlenose dolphins derived by Evans and Teilman (2009).

3.1.1.4 White-beaked dolphin

In the eastern North Atlantic, the ASCOBANS-HELCOM workshop (Evans and Teilmann 2009) found evidence for considering white-beaked dolphins from the northernmost part of Norway as a distinct MU but noted that individuals from all Norwegian coastal areas (north to south) appear to form a continuous and differentiated population that may be considered as a single separate MU, although more sampling in the southern coastal areas of Norway is necessary to corroborate this (Figure 3.7). The data suggest one continuous population within UK and Irish waters, and therefore individual white-beaked dolphins belonging to this area could be considered as a distinct MU. Photo-identification had also revealed matches between Scottish waters and the Danish North Sea and Skagerrak (Kinze, 2009).

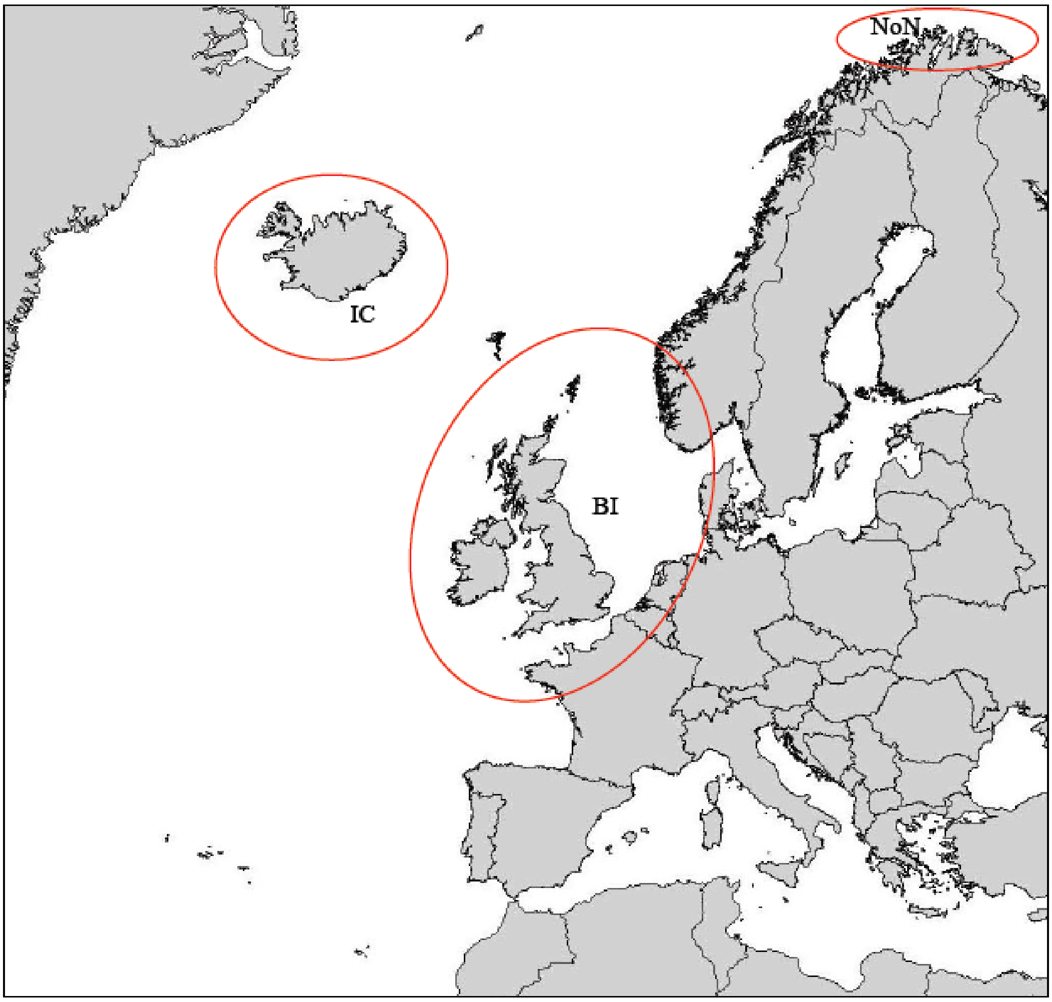

Figure 3.7. ASCOBANS recommended management units for white beaked dolphin (taken from Evans and Teilmann, 2009). IC – Iceland, BI – British and Irish waters, NoN – northern Norway.

Based on the available information, WGMME **endorses** three MUs for white-beaked dolphins in the eastern North Atlantic: a) northern Norwegian coast, b) waters around Britain and Ireland, and c) waters around Iceland.

3.1.1.5 White-sided dolphin

The ASCOBANS-HELCOM workshop proposed at least four management units for the white-sided dolphin in the North Atlantic but noted that these units may change if the number of sampling regions is increased (Figure 3.8). The four Management Units proposed were: a) northeastern North Atlantic including the northern North Sea; b) Central eastern North Atlantic including the Celtic Sea and western English Channel; c) Gulf of Maine; and d) Cape Cod. Further genetic analysis was considered necessary to corroborate the existence of two management units along this eastern seaboard.

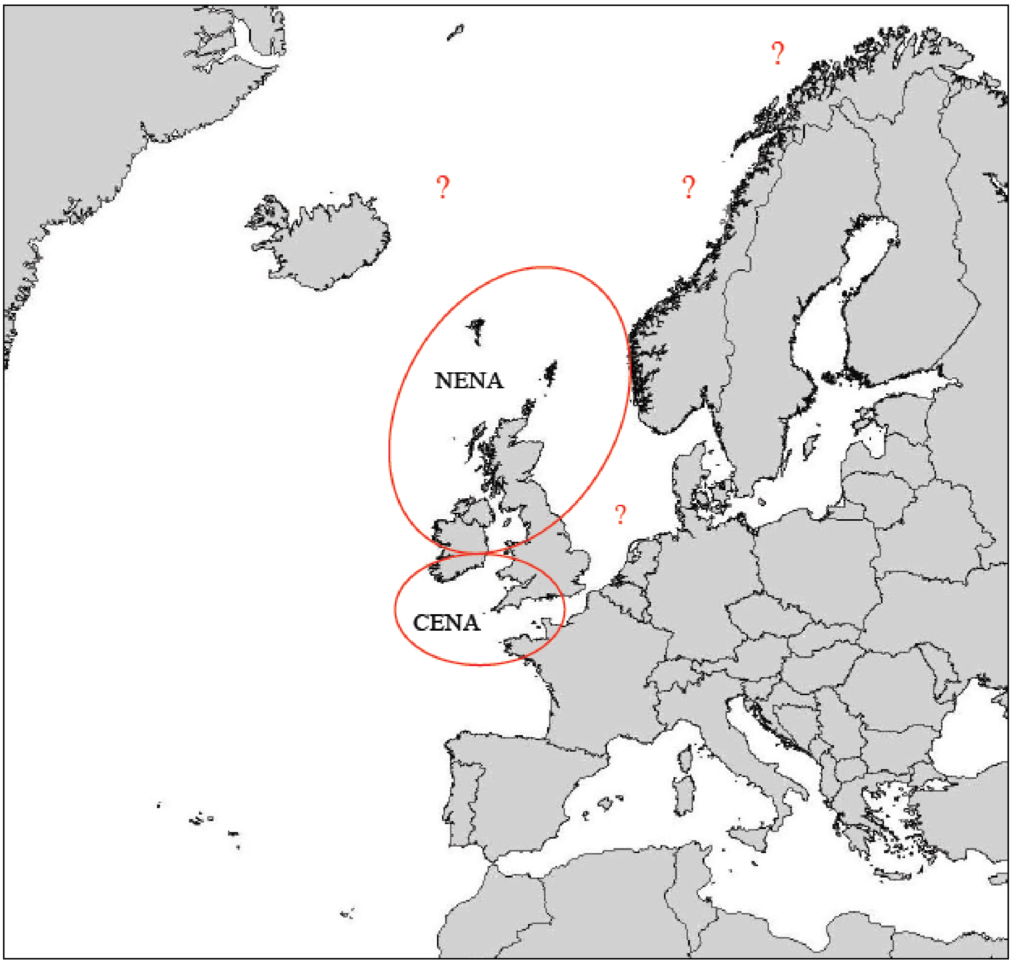

Figure 3.8. ASCOBANS recommended management units for white sided dolphin (taken from Evans and Teilmann, 2009). NEA – Northeastern North Atlantic including the northern North Sea, CENA - Central eastern North Atlantic including the Celtic Sea and Western English Channel.

Banguera-Hinestroza *et al.* (2010) analysed genetic variation at the mitochondrial (mtDNA) control region for 344 white-sided dolphin samples from three putative populations in the western North Atlantic and eight different regions in the eastern North Atlantic. The analyses showed high haplotypic diversity (Hd) at mtDNA (0.927 ± 0.007), but relatively low nucleotide diversity (0.00891 ± 0.0003). These findings suggest a pattern of genetic diversity congruent with an ancient bottleneck followed by an expansion in range in most *L. acutus* populations that were analysed. Population structure analyses showed that samples from the western region of the eastern North Atlantic (West Ireland, Faroe Islands and northwest British Isles) were similar to samples from the western North Atlantic (USA coasts). However, samples from the North Sea and eastern Scotland did show some degree of differentiation from other populations, from both the eastern and the western North Atlantic.

Mirimin *et al.* (2011) investigated nuclear and mitochondrial genetic variability of 42 Atlantic white-sided dolphins that stranded from 1990 to 2006 in county Mayo, Ireland, using eight microsatellite loci and 599 bp of the mitochondrial DNA control region. Results from both classes of markers were consistent with the hypothesis of a large random-mating population of white-sided dolphins off the northwest coast of Ireland. In addition, the analyses of two live mass stranding events (19 and five individuals, respectively) revealed that dolphins within each group were mainly unrelated to each other, suggesting dispersal of both sexes from the natal group (i.e., no natal phylopatry). Parentage analyses allowed the identification of mother–offspring pairs but ruled out all adult males as possible fathers. In combination with data on age of individuals, these results confirmed previous knowledge on life-history parameters, with sexually mature females ranging between 11 and 15 years of age and an inter-birth interval of at least two years.

The evidence for separation of the eastern North Atlantic into more than one MU is weak. WGMME **considers** that at this stage only one MU is necessary in the eastern North Atlantic.

3.1.1.6 Minke whale

The minke whale is widely distributed in the North Atlantic and is frequently observed along Icelandic and Norwegian coasts, as far south as Portuguese coasts. In eastern Canada, North Sea and Greenland and around Jan Mayen and Svalbard Islands. The population structure of minke whales in the North Atlantic has been investigated extensively as part of the process of developing the implementation of the IWC Revised Management Procedure (RMP) for this species in this region (see IWC 2009 for the most recent review). All information comes from animals in their summer feeding areas. There is no information from animals in their breeding areas, nor is it known where these breeding areas are.

The population structure used for management is for three "stocks" in the North Atlantic: Western stock (including Canada and West Greenland), Central stock (including East Greenland and Iceland) and eastern stock (including Norway). Each stock is divided into a number of "substocks". Each substock has its own preferred feeding or "home" subarea based on a range of biological and operational information. Uncertainties in the appropriate location of the boundaries of these subareas are taken into account by "mixing matrices" that define in which subareas (in addition to its "home" subarea) each substock could feed and the proportion of each substock that would be found in each subarea.

The RMP defines Small Areas as the Management Units in which catch limits are determined. Small Areas are defined as areas small enough to contain whales from only one stock, or to be such that if whales from more than one stock were present catching operations would be unable to harvest them in proportions different to their relative abundance in the area. The way in which the RMP is implemented for each species/region is reviewed approximately every five years through the Implementation Reviews.

At the 2003 Implementation Review (IWC 2004), new analyses continued to show that genetic differentiation was greater between the three putative stocks (eastern, central, western) than within them; data to assess structure within these stocks were available only for the eastern stock at the time. Genetic differentiation among eastern subareas was generally low, but was statistically significant in several cases, including between the North Sea and the area immediately to the north, although not between the North Sea and areas further north (Andersen *et al.*, 2003). Circumstantial evidence from pollutant levels, isotope ratios and fatty acid analysis was consistent with a distinction between the North Sea and other areas in terms of feeding (e.g. Born *et al.*, 2002; Born *et al.*, 2003). There was a significant genetic difference between the Barents Sea and areas to the west, which was maximised by a boundary at 28°E. No genetic evidence was found to support a distinction between the Vestfjorden area (EC Small Area) and surrounding areas. No significant genetic differences were found between the ES Small Area and areas to the south but there were operational considerations favouring retention of this area as a management area.

At the 2008 Implementation Review (IWC 2009), new genetic analyses found little evidence of population structure either between or within the central and eastern stock areas. Nevertheless, the same stock and substock structure were maintained for the purposes of implementing the RMP.

In a recent study, Anderwald *et al.* (2011) used microsatellite DNA and mtDNA markers to investigate minke whale population structure across the North Atlantic assessing the possible impacts of migratory behaviour on stock structure. No evidence of geographic structure among putative populations was found in the IWC management areas indicating that the minke whales of the North Atlantic were likely to be a single genetic population. However, using individual genotypes and likelihood assignment methods, two putative cryptic stocks were identified, which were independent of geographic location, i.e. they distributed across the North Atlantic in similar proportions in different regions (Figure 3.9). This supports the notion that individuals from different breeding populations form mixed assemblages at other times of the year. It was suggested that some differences found in the proportional representation of these populations may explain some of the apparent differentiation among regions detected in previous studies.

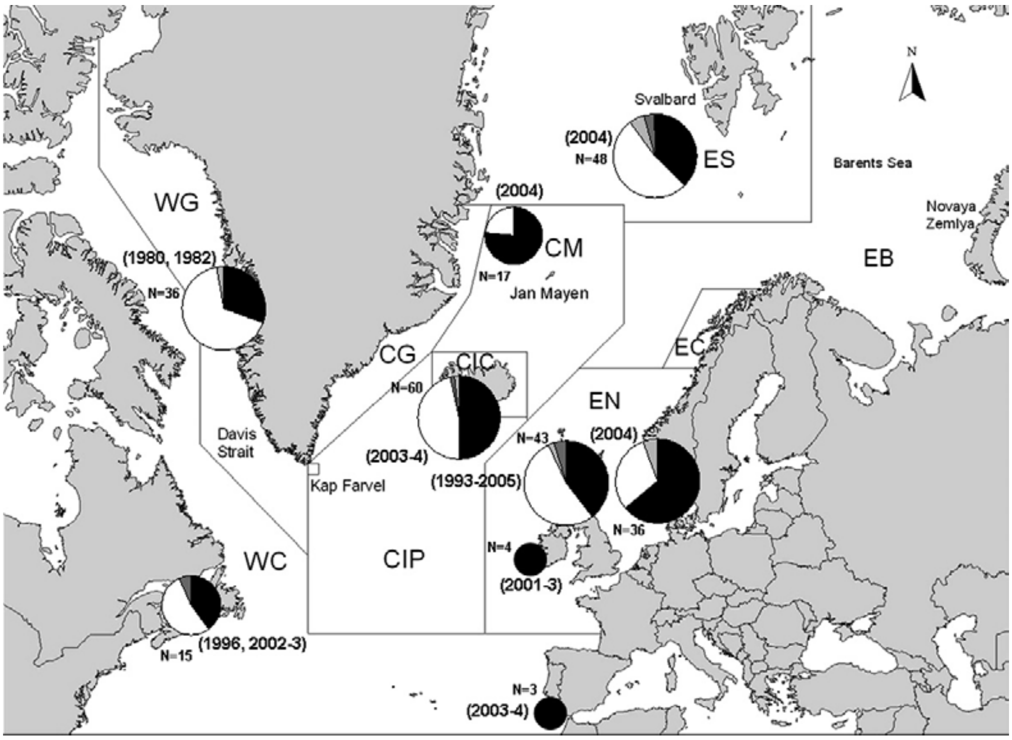

Figure 3.9. Sample sites in the North Atlantic within IWC management areas (West Greenland (WG), Central Eastern Greenland (CG), Central Jan Mayen (CM), East Svalbard (ES), East Barents Sea (EB), East Coastal Norway (EC), East North Sea (EN), Central Iceland Coastal (CIC), Central Iceland Pelagic (CIP) and West Canada (WC)). One sample site is included within a given management area, with the exception of EN, for which there were three sample sites: UK, Norway and Ireland. The geographic distribution of PBS1 and PBS2 in the North Atlantic according to GeneClass2 assignments of microsatellite genotypes is shown as pie charts: Black = PBS1, white = PBS2, light grey = putative PBS1 individuals assigned to PBS2, dark grey = putative PBS2 individuals assigned to PBS1. Sizes of pie charts indicate relative sample sizes for different areas. Sampling dates are given in parentheses. Taken from Anderwald *et al.*, 2011.

Data from the SCANS II survey indicated that during summer individuals in a subsection of the East and North Sea management area occurred in two areas of higher density, one in the North Sea and another off southern Ireland (Figure 3.10). The results of Anderwald *et al*. (2011) appear to suggest that individuals off Ireland likely belong to PBS1 whilst those in the North Sea are a mix of the two population types.

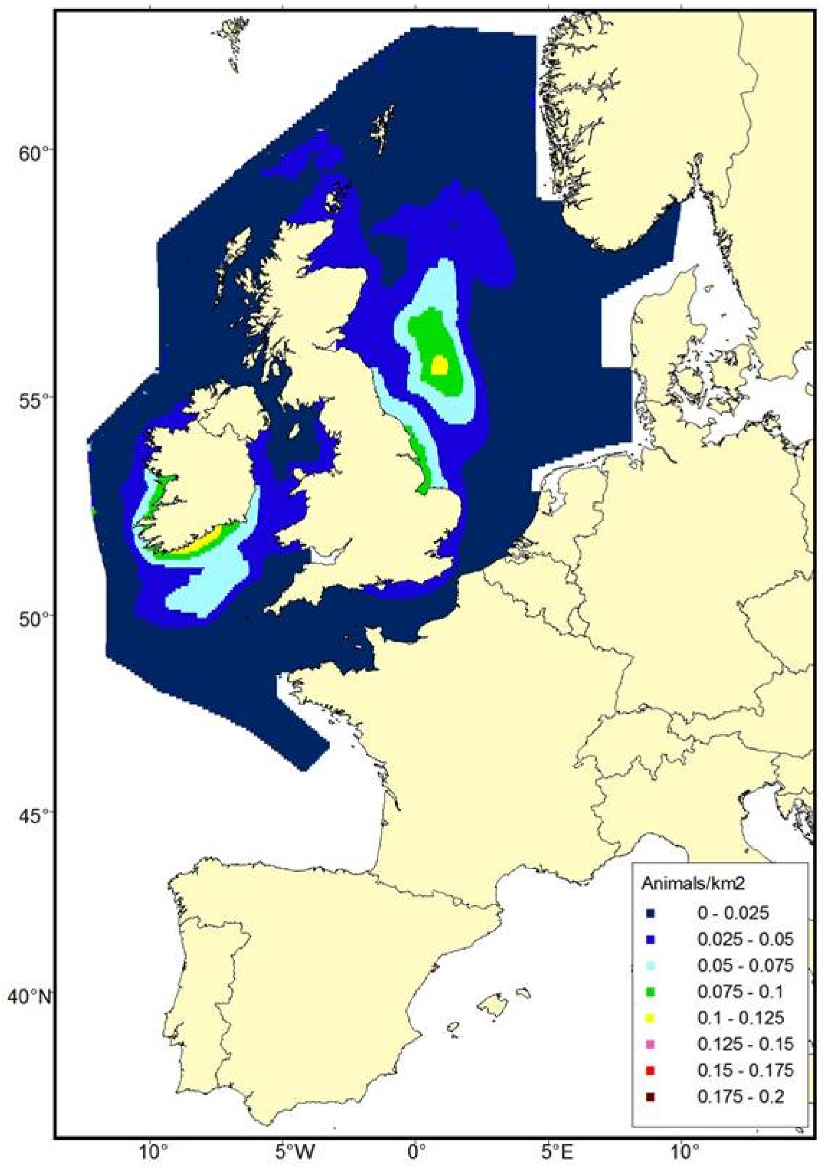

Figure 3.10. Minke whale density surface from SCANS II 2005 (Hammond *et al*., in prep.)

WGMME **recommends** that the management units proposed by the IWC are used for minke whales in the North Atlantic are retained at this time. The European North Atlantic group, therefore, comprises a single MU.

3.2 Overarching WGMME recommendation on MUs

WGMME **strongly recommends** that Member States use the proposed management units for reporting requirements of the Habitats Directive and for the development of indicators and their assessment for the Marine Strategy Framework Directive. In summary, there is a single MU in European North Atlantic for common dolphin (*Delphinus delphis*), white beaked dolphin (*Lagenorhynchus albirostris*), white sided dolphin (*Lagenorhynchus acutus*) and minke whale (*Balaenoptera acutorostrata*). For bottlenose dolphin (*Tursiops truncatus*) there are ten separate units closely associated with the mainly resident inshore populations in the European North Atlantic and a separate MU for the wider ranging mainly offshore animals. For harbour porpoise (*Phocoena phocoena*), MUs are proposed for the Iberian Peninsula, Bay of Biscay, Celtic Sea (including SW Ireland, Irish Sea and Western Channel) and NW Ireland/West Scotland and the North Sea. The MUs for harbour porpoises may need to be revisited as indicators for MSFD become better defined and aligned with ICES rectangles to enable the calculation of more accurate bycatch estimates. For the purposes of MSFD, it maybe that consideration of the species will need occur at the regional seas level (e.g. North Sea).

3.3 New survey and abundance information

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

Gilles *et al.* (2012) 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. The survey design comprised eight strata within the 66 768 km^2 study area. On 74 parallel transects planned, a total of 5997 km survey effort was carried out in good survey conditions during ten survey days between 28 July and 1 September 2011. In total 711 sightings with 1104 individuals were recorded, including 97 calves. The highest encounter rates were found in UK and Danish/German waters (Figure 3.11). Average porpoise density in the entire study area was estimated to be 1.82 animals km^{-2} (CV=0.31). Highest porpoise density was estimated for the western and northeastern part of the survey area whereas relatively low densities were estimated over the sandbank itself and to the southeast (Figure 3.12). The number of sightings of other species was too low to estimate density (seven minke whales, eleven white-beaked dolphins and 15 seals), all of them distributed on the slopes of the Dogger Bank. Harbour porpoises were also mostly found on the slopes of the bank, supporting the hypothesis that the higher biological production on the slopes attracts top predators.

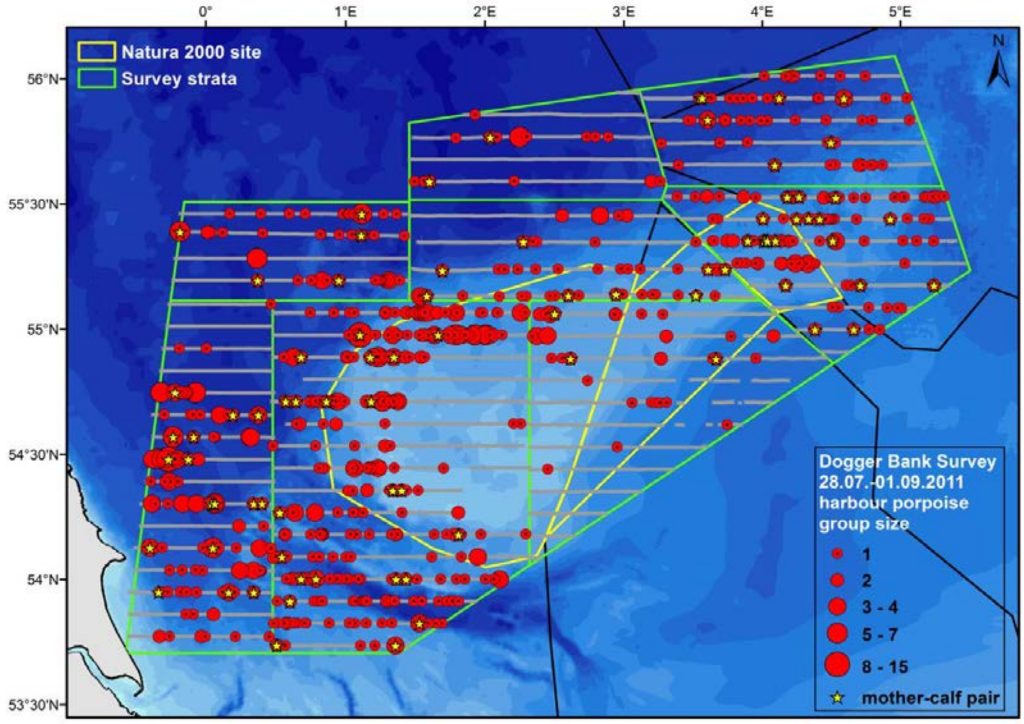

Figure 3.11 Realised effort (grey lines) and harbour porpoise group sightings. Only effort in good and moderate sighting conditions is shown (from Gilles *et al.*, 2012).

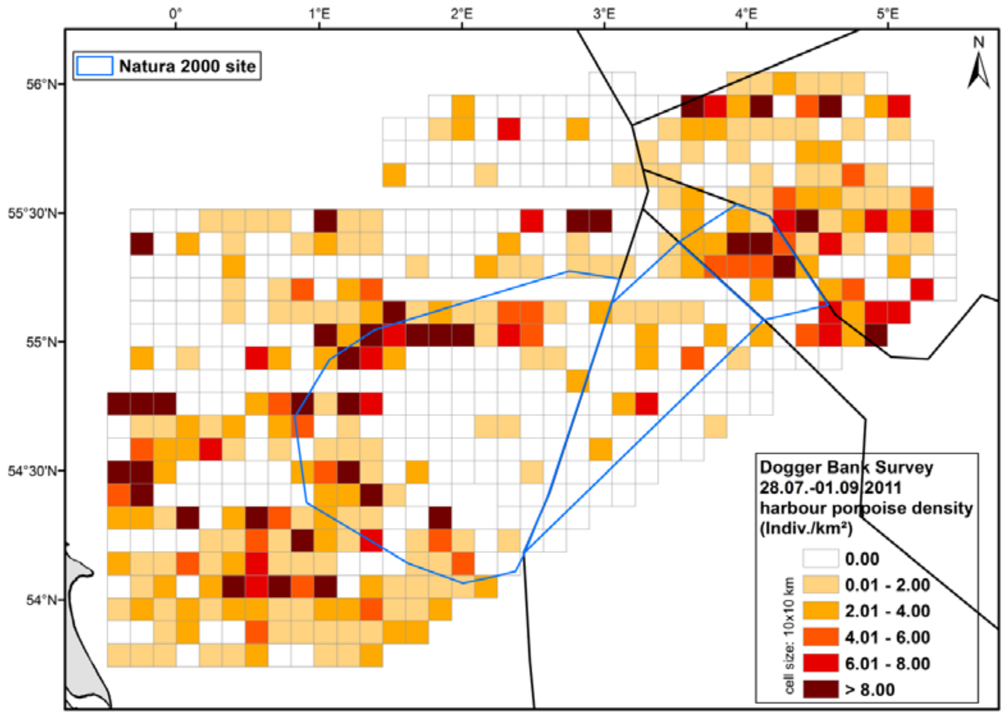

Figure 3.12. Spatial distribution of harbour porpoise density (indiv./km²) during the survey at the Dogger Bank in summer 2011. Grid cell size: 10x10 km. (From Gilles *et al.*, 2012).

3.3.2 Abundance of harbour porpoises 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 were conducted in the German North Sea and western Baltic Sea between May 2010 and August 2011 (Gilles *et al*., 2011). In the German North Sea, in the area of SCI *Sylt Outer Reef* (area C_Nord, Figure 3.13), harbour porpoise density was estimated to be 2.12 ind. km^{-2} (CV=0.31) in June 2010. A significantly lower density was estimated in July 2010 (0.88 ind km^{-2}, CV=0.33); the lowest summer density ever estimated since the beginning of the German surveys in 2002 (see Gilles *et al*., 2009). However, in June 2011 due to very high sighting rates (449 sightings in 5 hours), porpoise density increased to 4.75 ind. km^{-2} (CV=0.35) which is significantly higher than the estimate in June 2010. Similar to 2010, density in July 2011 was significantly lower than in June 2011. Density in the area C_Nord (and hence in the SCI *Sylt Outer Reef*) still belongs to the highest densities in the German North Sea and the highest number mother-calf pairs are sighted within that area (e.g. 102 calves in June 2011). During the survey in area D (incl. SCI *Borkum Reef Ground*, Figure 3.13) in March 2011, 126 sightings with 141 harbour porpoises were recorded, mainly west of the island Langeoog. Density was estimated to be 1.06 ind. km^{-2} (CV=0.38).During the survey in May 2011 332 sightings with 357 porpoises were recorded; most animals were sighted north and west of Borkum. In May 2011 density was estimated to be 1.59 ind. km^{-2} (CV=0.35). In comparison with earlier surveys conducted in Area D since 2002, the density estimated for March and May 2011 belong to the highest for that area. This indicates an ongoing increase of porpoise density in the southeastern North Sea.

In the southwestern Baltic Sea four aerial surveys were accomplished during short time frames in spring, summer and autumn 2010 as well as in spring 2011. In May 2010 only 26 sightings with 29 harbour porpoises were recorded. Porpoises were only sighted in the west of the study area where density was estimated to be 0.38 ind. km^{-2} (Kiel Bight, block E, Figure 3.13). In July 2010, 62 sightings with 75 porpoises were recorded; these predominantly in the north and west of the island of Fehmarn. A density of 0.37 ind. km^{-2} was estimated for the entire study area. In September 2010 38 sightings with 47 harbour porpoises were recorded. Many porpoises were sighted in the Kiel Bight, the Flensburg Fjord, north of the Danish islands Als and Ærø and to the east of Fehmarn and the Mecklenburg Bight. For the entire area the density was estimated to be 0.23 ind. km^{-2}. In June 2011, 33 sightings with 38 porpoises were recorded. Relatively few sightings were recorded in the Kiel Bight and the density in the western Baltic Sea was estimated at 0.27 ind. km^{-2}. Compared with surveys conducted since 2002, densities are decreasing in the Kiel Bight since May 2010. Porpoise density in the Fehmarn Belt area and Mecklenburg Bight varies strongly since 2002; it is, however, rarely higher than 0.3 ind. km^{-2} (Gilles *et al*., 2011).

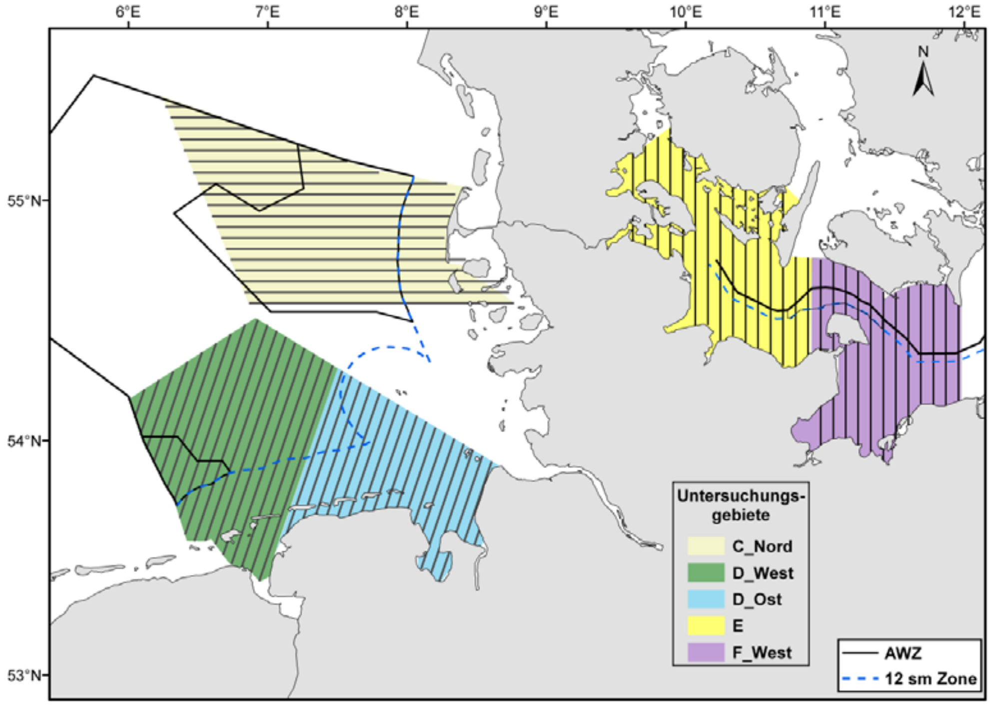

Figure 3.13. Survey blocks in the North and Baltic Sea. Parallel transects are spaced 5 km apart in the North Sea and 6 km in the Baltic Sea. From Gilles *et al.*, 2011.

3.3.3 Abundance of harbour porpoises in Dutch waters

Between May 2008 and March 2011, aerial surveys were conducted in Dutch waters (Geelhoed *et al.*, 2011; Scheidat *et al.*, 2012). The aim of these surveys was to assess the seasonal abundance and distribution of harbour porpoises *Phocoena phocoena* on the Dutch Continental Shelf (DCS), and how their distribution varies in space and by season. Three complete aerial surveys of the DCS were conducted along predetermined track lines, in summer (July 2010), late autumn (October/November 2010) and early spring (March 2011) (Figure 3.14). In total 1085 sightings (1236 animals) of harbour porpoises were recorded, five sightings of white-beaked dolphins *Lagenorhynchus albirostris* (eight animals) and 64 sightings (66 animals) of grey *Halichoerus grypus* and harbour seals *Phoca vitulina*. Mother–calf pairs of porpoises were mostly sighted in July, around and west of the wind farm survey Area W1, suggesting that porpoises reproduce in Dutch waters. The data was analysed with standard distance sampling methodology. The resulting density estimates of harbour porpoises for the DCS were 0.44 animals/km² in July, 0.51 animals/km² in October/November and 1.44 animals/km² in March. This means total numbers for the entire DCS of ca. 26 000 animals in July (95% Confidence Interval (C.I.): 14 000–54 000), ca. 30 000 in October/November (C.I.: 16 000–59 000) and ca. 86 000 in March (C.I.: 49 000–165 000). These numbers represent a substantial part of the population where the Dutch porpoises belong to, the so-called management unit southwestern North Sea and the Eastern Channel. Based on the SCANS II data from 2005 the estimated number of porpoises in this management unit is less than ca. 180 000 animals.

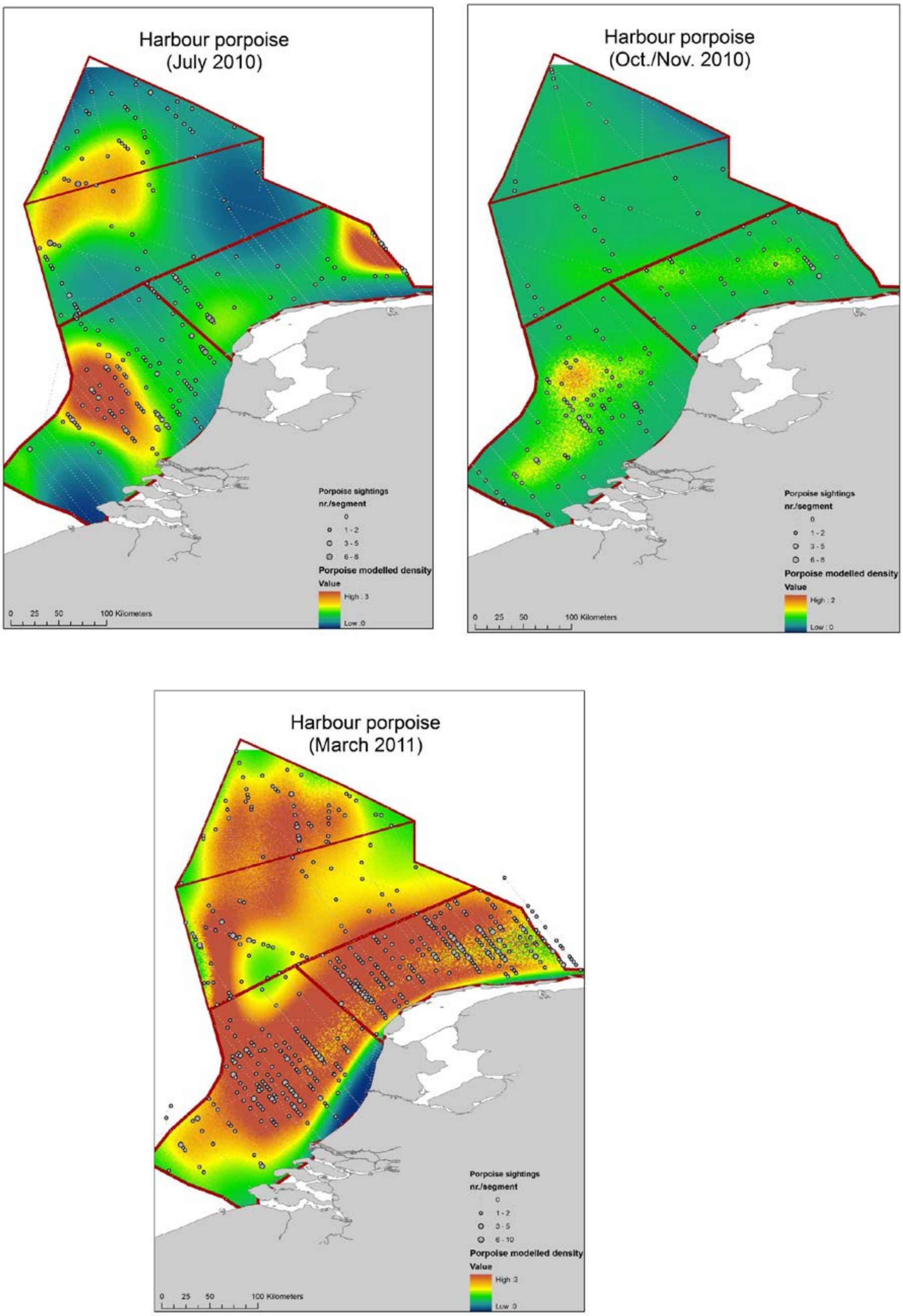

Figure 3.14. Estimated distribution of harbour porpoises (animals/km2), July 2010, October/November 2010 and March 2011. Predictions were made for sea state 2, turbidity 1, cloud cover 4/8 and for 1 PM UTC. (Taken from Goelhoed *et al*., 2011 and Scheidat *et al*., 2012).

3.3.4 Harbour porpoise in the English Channel

Cucknell *et al*. (2012) undertook a visual and acoustic survey of the English Channel for harbour porpoises between May and June 2011. A total of 4243 km track line was completed, with 2749 km "on track" with acoustic effort. Visual effort was impacted by poor sighting conditions due to the weather. Forty encounters with cetaceans occurred during the survey (16 visual and 24 acoustic), 34 of which were for harbour porpoise (13 visual and 21 acoustic, with three detections coinciding). Most of these occurred in the western channel area in depths of 50–100 m (Figures 3.15 and 3.16).

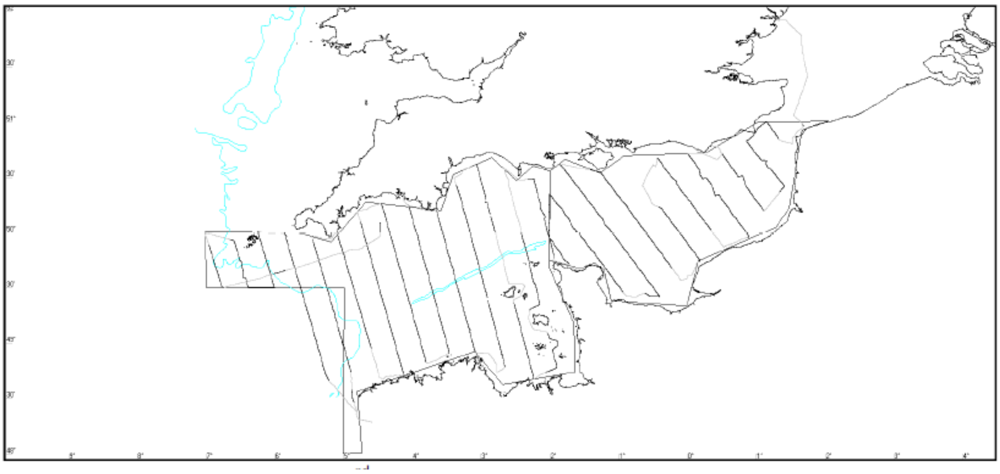

Figure 3.15. Survey effort. Distance logged was 4243 km of which 2749 km was on track with acoustic effort (From Cucknell *et al.*, 2012).

Figure 3.16. Harbour porpoise detections (stars - definite detections, triangles – possible detections) (From Cucknell *et al.*, 2012).

3.3.5 Abundance of common dolphins in Portuguese waters

Since 2010 Portugal has established a monitoring programme concerning distribution and abundance estimate of marine mammals. This monitoring scheme is supported by SafeSea (EEAGrants) and MarPro (Life+) projects and will continue systematically until 2015. Presently, the aerial census allowed for a correct analysis of distribution and abundance of common dolphin only (Figures 3.17 and 3.18) indicating an abundance estimate of 20 500 individuals (Table 3.1).

Table 3.1. Summary of abundance of common dolphin estimated during the SafeSea project.

Species	Sightings/km2	Sightings/km2 CV	Sightings/km2 95% CI	Animal/km2	Animal abundance (all Portugal, from the coast till 50 nm)
Common dolphin	0.0229	0.29	0.0135–0.0389	0.263	20 500

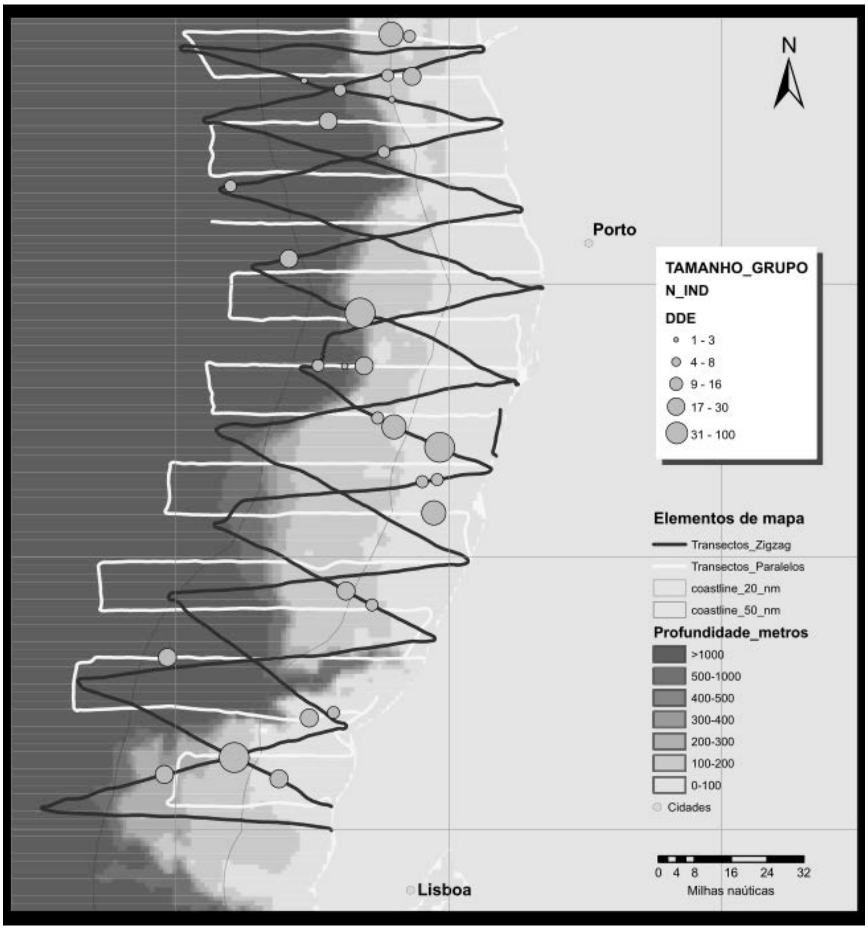

Figure 3.17. Distribution of sightings of common dolphin during aerial surveys.

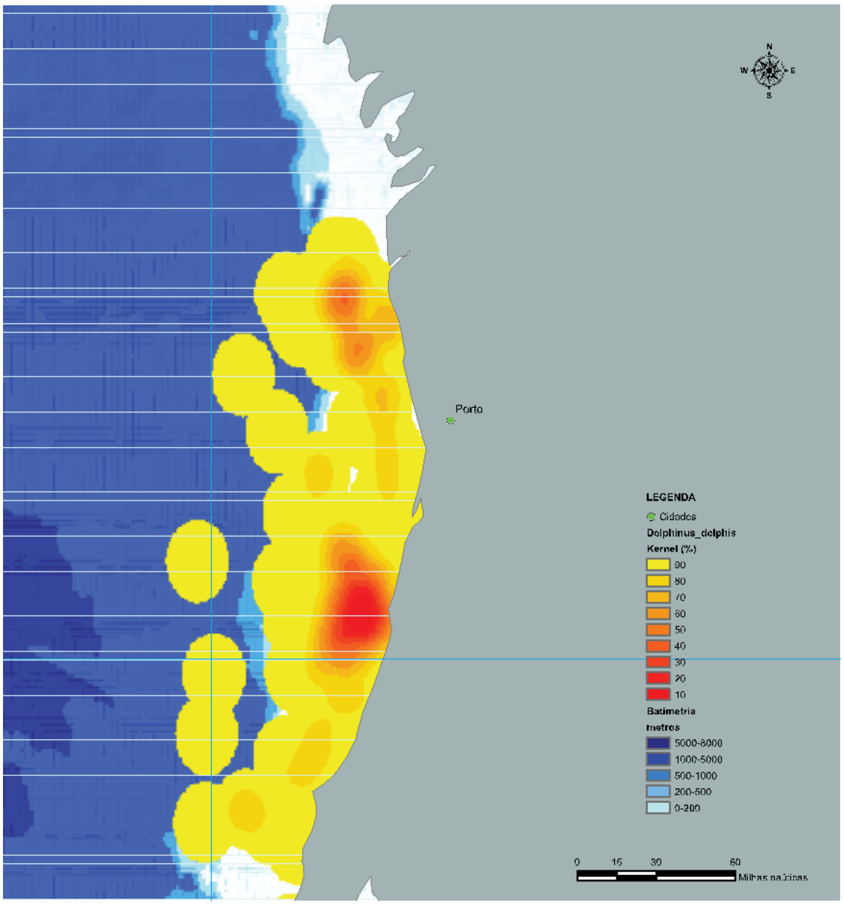

Figure 3.18. Distribution of common dolphin using all information collected during SafeSea project (representation of Kernels).

3.4 Interpreting numbers of stranding events

Recent developments were aimed at improving the monitoring value of stranding data by constructing a framework for the interpretation of stranding datasets (Peltier *et al.*, 2012) and proposing several spatial indicators (Peltier, 2011). By using the drift model MOTHY (*Mobilité des Hydrocarbures*) initially developed by *MétéoFrance* to predict the drift of oil slicks and later adapted to floating objects in the context of maritime safety, it was possible to model the drift of cetacean carcasses (Peltier *et al.*, 2012). The immersion rate, which is a central parameter for the MOTHY model, was experimentally determined for two small cetaceans, as well as visual criteria allowing the duration of the drift to be estimated. Dead common dolphins were tagged and released at sea from fishing vessels, allowing apparent stranding rate to be estimated at 8%; the other 92% were partly explained by the effective stranding rate (only 56% predicted to reach a coast within 40 days post-mortem) and by the combined probabilities to sink after death and to be unreported after stranding. Then, 238 theoretical dolphins regularly spaced from northern Spain to southern Norway all the way across the Bay of Biscay, Channel and North Sea were simulated to drift in order to predict their stranding. Model runs were conducted every ten days over the period 1990–2009 resulting in maps of stranding probability averaged by months, seasons or the whole year; in addition, prediction of stranding (locations and dates) under the null hypothesis were produced (here, H_0 means that cetaceans and mortality are uniformly distributed in space and time). Finally, real stranding datasets of harbour porpoise and common dolphin gathered from stranding schemes of Belgium, France, the

Netherlands and the UK were used to back calculate their origin with MOTHY. Comparisons between the null hypothesis and stranding observation reveal anomalies that are the difference between expected and observed stranding datasets. Hence, anomalies of stranding time-series or seasonal patterns can be generated, as well as anomalies of stranding distributions along the coasts and anomalies of mortality at sea. These anomalies represent departures from a flat theoretical distribution in abundance and/or mortality.

At this stage, this interpretation framework for stranding datasets is still in need of further trials, refinements and developments. In particular, extending the calculation area to the west of the Iberian Peninsula and of the British Isles would be a priority. Analyzing stranding anomalies at fine spatio-temporal scales would help resolving issues related to variations of reporting effort, and proposing correction factors. Using a real cetacean distribution (*e.g.* a density surface model), instead of a flat theoretical distribution to predict stranding, would allow one to assimilate anomalies to mortality and therefore identify hot-spots, either spatial or temporal, of mortality. Comparing these anomaly maps and time-series to statistics on human activities (e.g. fishery statistics) would help flagging situations of concern in terms of conservation. Finally, the use as reference sectors of some specific stretches of the coastline where all stranded carcasses are mechanically collected would help disentangling the probabilities that a dead cetacean would float and that a stranded carcass would be reported. Once this is done, stranding datasets would have the potential to infer cetacean mortality, possibly broken down by death causes, within a known surface area of the ocean.

3.5 Large scale cetacean surveys in the European Atlantic

3.5.1 Abundance estimates from CODA and SCANS-II

WGMME (2009) presented estimates of abundance from the Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA) survey in 2007 and WGMME (2010) presented density surface modelled maps of distribution from the same data. Last year, WGMME (2011) reported that updated abundance estimates from an analysis of combined data from SCANS-II, CODA and the Faroes block of T-NASS would become available in the coming year. Recently discovered minor issues with the processing of the SCANS-II and CODA data are currently being addressed, which will alter the abundance estimates very slightly. The final abundance estimates from the combined analysis will be available later in 2012.

3.5.2 Abundance estimates from T-NASS

The NAMMCO Working Group on Abundance Estimation met in March 2011, primarily to review and finalise as many estimates of abundance as possible from T-NASS 2007. The current complete set of agreed estimates is given in Table 3.2, which replaces Table 1 given in WGMME (2011).

Table 3.2. Abundance estimates from T-NASS (2007), and additional estimates, endorsed by NAMMCO.

Survey Areas	West Greenland	Iceland Coastal (Faroese coastal)	Iceland-Faroes	Canada GSS	Canada NL	Norwegian mosaic 2003-2007
Species / Survey	Aerial	Aerial	Shipboard	Aerial	Aerial	Shipboard
Fin whale	4359 n, j (1879–10 114)		20 613 n, j (14 819–25 466) 26 117 p, j (17 401–39 199)	462 n, j (270–791)	1254 p, j (765–2059)	To be done
Minke whale	16 609 p, a[1], j (7172–38 461) 22 952 p, a[2], j (7815–67 403)	14 638[3] p, a, l (7381–24 919) 20 834[4] p, a, l (9808–37 042)	10 782 n, k (4733–19 262)	1927 j (1196–2799)	3748 p, j (2214–6345)	IWC
Minke whale (2009)		9588 p, a, l (5274–14 420)				
Humpback whale	3272 p, a, j (1.230–8.710)	1242 p, j (632–2445)	11 572 n, j (4502–23 807)	653 j (385–1032)	3712 p, j (2536–5428)	To be done
Pilot whale	**2976 n, j (1178–7515)**			6134 n, j (2774–10 573)		To be done
Sperm whale			To be done			To be done
Bottlenose whale			To be done			To be done
Harbour porpoise	**33 271 p, a, j (15 939–69 450)**	**43 179 p, a, l (31 755–161 899)**		3667 n, j (1565–6566)	958 n, j (470–1954)	To be done
Harbour porpoise Faroes (2010)		**5175 p, a, l (3457–17 637)**				
White-beaked dolphin	**9827 p, j (6723–14 365)**	To be done	To be done			To be done
White-sided dolphin			To be done	4289 n, j (cv = 0.210)	3,086 p, j (1,781-5,357)	To be done
Common dolphin				53 049 n, j (34 865–80 717)	613 p, j (278–1355)	

Estimates in bold are the first estimates for the species in the area.

n - uncorrected for bias; p - corrected for perception bias; a - corrected for availability bias

[1] Availability bias adjusted using aerial photographic images taken in Iceland

[2] Availability bias adjusted using satellite tagging data from three different areas

[3] Using both primary observers

[4] Using only the most effective primary observer (much higher sighting rate)

i - Endorsed at the NAMMCO WG on Abundance Estimation, Copenhagen, April 2008, and subsequent Scientific Committee Meeting (NAMMCO, 2009)

j - Endorsed at the NAMMCO WG on Abundance Estimation, Quebec, October 2009, and subsequent Scientific Committee Meeting (NAMMCO, 2011)

k - Endorsed at the NAMMCO WG on Assessment, Copenhagen, March 2010, and subsequent Scientific Committee Meeting (NAMMCO, 2011)

l - Endorsed at the NAMMCO WG on Abundance Estimation, Copenhagen, March 2011

3.5.3 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 have begun to initiate the development of a project for a large scale survey to estimate the absolute abundance of cetaceans in European Atlantic waters to follow on from SCANS in 1994, SCANS-II in 2005 and CODA in 2007. The current target date for such a survey (SCANS-III) is 2015, 10 years after SCANS-II. The intention is that shelf and offshore waters will be covered. To achieve this it will be necessary to consider the balance of shipboard and aerial survey and to pursue some technical development in aerial surveys; the latter will be discussed at an ECS workshop later this month. Aerial surveys are now being considered for the majority of the proposed cetacean survey in the Mediterranean co-ordinated by ACCOBAMS and will continue to be used in a second Trans-North Atlantic Sightings Survey (T-NASS) also proposed to take place in 2015 (ACCOBAMS, 2010; NAMMCO, 2012). Coordination of CODA and T-NASS worked well in 2007 and will continue for the 2015 surveys.

WGMME strongly supports the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2015 and **recommends** that it is supported by all range states and by ICES, ASCOBANS and the European Commission. Continuation of these surveys is essential for the accurate calculation of absolute abundance estimates that are required for reporting of the Habitats Directive and the Marine Strategy Framework Directive.

3.6 Population structure of harbour seals in the eastern North Atlantic

The population structure of harbour seals (*Phoca vitulina*) around the UK, with a particular focus in Scotland, has been investigated using genetic markers to investigate population structure (Islas-Villanueva *et al*., 2012). The population genetic structure identified has been compared to recently defined harbour seal management regions in Scotland (SCOS, 2011), which have been defined based on data from haul-out and pupping sites and on the tendency for harbour seals to forage relatively close to haul-out sites, as inferred from telemetry data.

DNA was obtained from a total of 453 individuals around Scotland and from comparative regions in the rest of the UK and Europe (and including an out-group of Pacific harbour seals). Bayesian clustering analysis clearly separated Scotland from England, Normandy and the Dutch Wadden Sea. In the data from Scotland and

Norway, three clusters were identified: (a) Norway, (b) West Coast of Scotland + Northern Ireland and (c) Pentland Firth + Orkney + Shetland + Moray Firth + Firth of Tay with some degree of shared individuals between them.

Examining the Scottish populations alone indicated a subtle differentiation between the Tay Estuary and other north and east coast groups. Normandy showed strong substructuring within the samples analysed, with approximately half of the sample clustering with the Dutch Wadden Sea and England and the other half showing a different origin.

Allelic diversity (n) and heterozygosity (*Ho*) are standard measures that assess the level of inbreeding which populations display as a reflection of their 'genetic health'. The putative populations with relatively good sample sizes and low levels of genetic diversity were Shetland (n=2.545, *Ho=0.363*), the Outer Hebrides (n=2.467, *Ho= 0.331*) and Normandy (n=2.69, *Ho=0.341)*.

These results support the management regions defined for harbour seals in Scotland. Some broader genetic clustering is apparent, but 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.

3.7 North Atlantic right whale population structure

Silva *et al.* (2012) reported that a right whale (*Eubalaena glacialis*) from the western North Atlantic population, sighted in the Azores, was subsequently found to have moved back to the northwest Atlantic. The whale was sighted in the Azores on 5 January 2009 travelling in a west–south westerly direction at a constant speed. A photographic match was found to an adult female in the North Atlantic Right Whale Catalogue. The whale's previous last sighting, on 24 September 2008 in the Bay of Fundy, Canada, implies movement to the Azores of at least 3320 km in 101 days. It was subsequently resighted in the Bay of Fundy on 2 September 2009, 237 days after being seen in the Azores. This appears to be the only documented evidence of a western North Atlantic right whale outside its normal range in winter, and provides additional evidence of the potential for interbreeding between western North Atlantic right whales and the remnant eastern population.

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4 Tor b) Develop biodiversity indicators in support of policy drivers, and develop indicators that are robust to expected uncertainties in data and/or to provide a quantitative analysis of the potential effects of data limitations on indicator performance

4.1 Introduction

In 2010, WKMARBIO requested that WGMME begin to develop biodiversity indicators for marine mammals. At the meeting, WGMME discussed the strengths and weaknesses of current and proposed indicators types for marine mammals and discussed possible indicators that could be used for supporting policy drivers. It was decided that, in 2012, WGMME would focus on development of the proposed biodiversity indicators, ensuring that these were robust to expected uncertainties in data and/or provided a quantitative analysis of the potential effects of data limitations on indicator performance (**Action d,** WKMARBIO 2011 Report).

In the intervening period, OSPAR, which has a role in coordinating the implementation process for the Northeast Atlantic region, also convened a workshop on the development of indicators in support of the Marine Strategy Framework Directive (2008/56/EC). Within OSPAR, this work is overseen by the Intersessional Correspondence Group on the Marine Strategy Framework Directive (ICG-MSFD). The Intersessional Correspondence Group for the Coordination of Biodiversity Assessment and Monitoring (ICG-COBAM) is the main delivery group within the OSPAR framework for coordination in relation to the biodiversity aspects of the MSFD. The workshop was organized as part of ICG-COBAM's programme of work on MSFD, under the lead of ICG-MSFD.

The workshop terms of reference stated that its purpose was to undertake a comparison and analysis of indicators and associated targets for MSFD biodiversity descriptors 1, 2, 4 and 6 between OSPAR Contracting Parties also involved in the implementation of the MSFD and to identify where common indicators could be identified.

The workshop resulted in summary reports and detailed analyses per ecosystem component (including one on marine mammals) with proposed indicators, associated targets, relevance to different subregions and some preliminary agreement on species/metrics and targets. From the results it was concluded that there are some promising commonalities between proposed indicators, especially relating to abundance, distribution and bycatch of key species. During the workshop a number of actions were identified that would need to be undertaken in order to take forward the work started by the workshop. These actions relate to the facilitation of further expert discussions, the need for scientific research, and operationalisation of indicators for monitoring.

Subsequently, ICG-COBAM, as part of the development of the Advice Manual on MSFD indicators developed a series of summary sheets for the 'common' indicators (Annex 1). These were made available to WGMME for further consideration and development of indicators prior to their publication. For the purposes of WGMME 2012, the 'further needs' section of each of these summary sheets was focused upon. The WGMME also assessed other possible indicators that could be incorporate within the

MSFD. This should be taken under consideration by ICG-COBAM during their next round of assessments.

4.2 Development of seal indicators through ICG-COBAM

Common indicators proposed for grey and harbour seals by Member States where largely based on monitoring currently undertaken for OSPAR's seal EcoQOs. It should be noted that prior to the development of the EcoQOs, monitoring procedures or programmes for harbour and grey seal populations in the North Sea were already in place. These were designed to effectively assess the distribution and numbers of seals in different areas/countries (Reijnders *et al*., 2003; Thompson *et al*., 2005; Lonergan *et al*., 2007). The results of these monitoring programmes were used by OSPAR to develop the EcoQO for each species which were considered to provide a measure of seal performance. There was a general consensus from Member States that MSFD indicators for seals could be developed for:

1) **Criterion:** Species distribution. **Indicator:** Distributional range and distributional pattern within range. **Parameter/metric:** Distributional range and pattern of grey and harbour seal breeding colonies and haul-out sites, respectively.
2) **Criterion:** Population size. **Indicator:** Population abundance and/or biomass, as appropriate. **Parameter/metric:** Abundance of grey and harbour seal at breeding colonies and haul-out sites, respectively.
3) **Criterion:** Population condition. **Indicator:** Population demographic characteristics. **Parameter/metric:** Harbour seal and Grey seal pup production.
4) **Criterion:** Population condition. **Indicator:** Mortality rate. **Parameter/metric:** Numbers of individuals within species being bycaught in relation to population.

HELCOM have proposed seal indicators for population abundance in terms of growth rate as a percentage increase per annum and population condition assessments of blubber thickness and pregnancy rates in grey seals in the Baltic Sea (HELCOM Report, 2012a, 2012b). No indicators were proposed for distributional range, or distributional pattern within range.

The seal monitoring currently undertaken in the European North Atlantic is based on pup production counts for grey seals and haul-out counts during the moult for harbour seals. OSPAR outlined that further information is required prior to the establishment of the proposed indicators, including:

1) Compilation of existing data on abundance and distribution of grey and harbour seals.
2) Subdivision of the area (beyond the North Sea) into management units.
3) Development of a baseline for each management unit.
4) Development of a standardized monitoring methodology, or alternatively a mechanism for standardizing data post collection.
5) Development of an assessment tool.

For bycatch considerations see Section 4.5. It should be noted that not all Member States have initiated seal bycatch monitoring programmes. The UK include seals within the UK Bycatch Monitoring Scheme which was developed in order to meet the needs of EU Regulation 812/2004 and Article 12 of the Habitats Directive. In Ireland, a pilot study has been initiated on the west coast to assess seal depredation and bycacth in inshore fisheries (M. Cronin, pers. comm.). The results of the pilot could inform on the development of a national bycatch monitoring programme. It is unclear at this time if any other Member States monitor the bycatch of seals. WGMME (2008, Section 4.3) '*noted that despite all of the observations made under EC Regulation 812/2004, there is little mention in national reports of any seal bycatch, and recommends to the European Commission that bycatches of seals and other protected species should be reported by observer programmes established under the 812/2004 regulation as well as those conducted under Data Collection Regulations for discard sampling*.' In recent years SG/WGBYC have periodically tried to assess the bycatch of seals but have not been able to do this due to the lack of suitable data.

4.2.1 OSPAR's seal EcoQOs

Many of the seal indicators put forward through OSPAR and ICG-COBAM were linked to the monitoring currently being undertaken which is used in the assessment of the two OSPAR Ecological Quality Objectives (EcoQOs) for seals in the North Sea. The OSPAR EcoQOs were developed to function both as indicators (to provide specific issues for monitoring) and objectives (against which to measure progress). As an entire group, the EcoQOs were intended to provide comprehensive coverage of the ecosystem and the pressures acting upon it, so that meeting all EcoQOs should indicate that the ecosystem is in a good state. Where EcoQOs are not met, it should trigger further investigations by Contracting Parties into possible reasons for the failure and, where changes are due to anthropogenic activity, indicates the need for appropriate measures to regulate the relevant human activity.

Monitoring of seals is undertaken on a site by site basis and the results combined to provide population estimates for the subunits listed. WGMME (2009; see Section 6) reviewed the seal subunits for the North Sea and suggested some changes to the EcoQO wording such that the stated subunits equate to those actually used for monitoring. The proposed changes were based on improved knowledge of seal movements from tags and photo ID studies, as well as genetic work. The term subunit is taken to be equivalent to the ICG-COBAM term 'management unit'. The term 'management areas' is also utilised to define regional breeding group of grey seal or haulouts for harbour seal, e.g. in the UK. Although these have evolved over time, the original management areas were used for the EcoQO subunits.

4.2.1.1 Harbour seal

The EcoQO for harbour seals states:

> *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. These subunits are: Shetland; Orkney; North and East Scotland; Southeast Scotland; the Greater Wash/Scroby Sands; the Netherlands Delta area; the Wadden Sea; Heligoland; Limfjord; the Kattegat, the Skagerrak and the Oslofjord; the west coast of Norway south of 62°N.*

In 2009, WGMME recommended the use of four harbour seal management units within southern Scandinavia waters in the North Sea area: (1) Skagerrak, (2) Kattegat,

(3) central Limfjord and (4) the Wadden Sea; therefore splitting the current EcoQO subunit Kattegat, Skagerrak and Oslofjord. Within the UK colonies, it was recommended that the North and East Scotland subunit be split with the north coast being included within Orkney group, and the east coast changed to Northeast Scotland (Moray Firth). This recommendation does not appear to have been acted upon by OSPAR to date.

The 2010 assessment of this harbour seal EcoQO (Figure 4.1), which covered the five years to 2006, was not met in several areas where declines of seals of more than 10% occurred (Shetland, Orkney, east of Scotland, Greater Wash to Scroby Sands, Limfjorden in Denmark and the west coast of Norway). Of the areas where declines were noted, the Greater Wash and Limfjorden areas had been affected by an outbreak of the morbilli virus. In other areas, the cause of the decline was unknown. It should be noted, however, that data used for the Wadden Sea covered a shorter period 2003–2006 and so only included the recovery period after the 2002 PDV outbreak when the population was increasing at a rapid rate.

Between 2001 and 2006, the harbour seal population in Orkney and Shetland declined by 40% (95% confidence interval: 30–50%), indicating harbour seals in these areas experienced substantially increased mortality or very low recruitment over this period (Lonergan *et al*., 2007). Little evidence of PDV was noted and there was no evidence of emigration to other UK colonies. The declines noted were more than four times the threshold of OSPAR EcoQO for this species in the North Sea. The proximate causes of a decline on the scale could only be a sustained high level of reproductive failure or increased rates of mortality, or some combination of these causes (Lonergan *et al*., 2007).

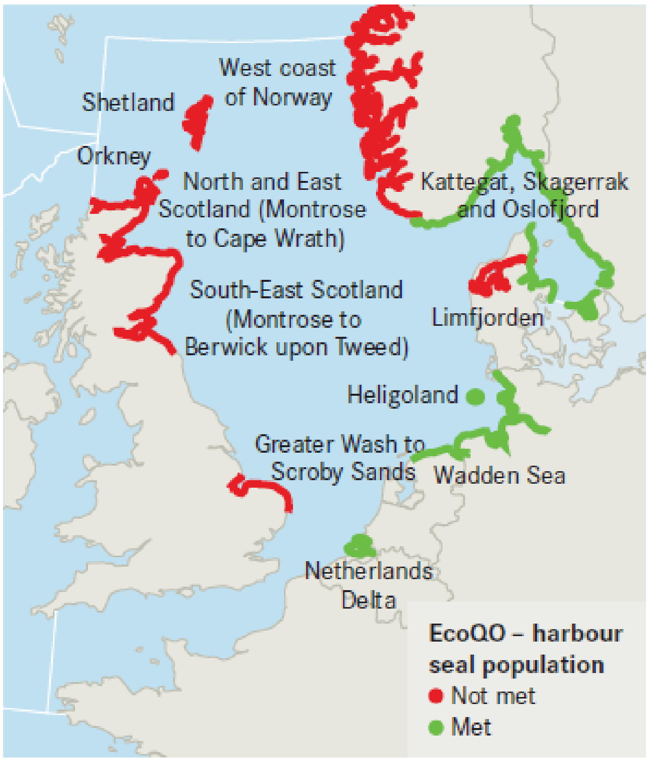


Figure 4.1. 2010 EcoQO assessments for harbour seal (from OSPAR, 2010).

Recent assessments from the Wadden Sea (2008–2011), indicate that harbour seal populations are continuing to increase, although the rate of increase has been declining possibly indicating the population is approaching carrying capacity (Trilateral Seal Expert Group, 2011a). Additionally, at the extreme south of the species distribution in Europe, haul-out counts have also been increasing at >20% yearly over the period 1993–2008 (Figure 4.2; Hassani *et al.*, 2010). This rate of increase is greater than that recorded into the Wadden Sea after the PDV outbreak when the population growth, averaging 12.7% per year was considered to be close to exponential over a 14 year period (Reijnders *et al.*, 2003). Thus, the increase in southern colonies is suggestive of an important element of immigration from other regions.

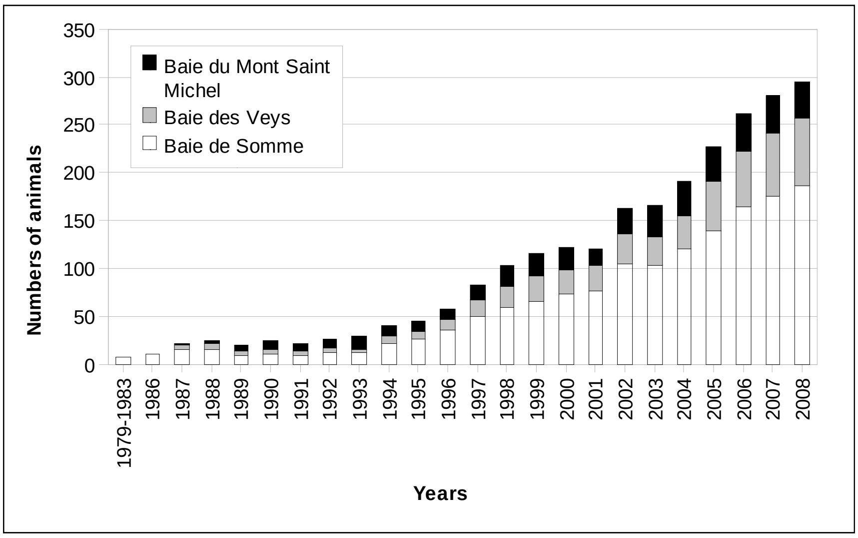


Figure 4.2. Increasing abundance of harbour seals at three French colonies.

Tracking data from harbour seals in the UK colonies does not appear to indicate movement from these colonies to the French ones (Figure 4.3). However, genetic analyses suggest that the harbour seals in French colonies are a mixture of animals from English and Dutch sites and others of unknown origin (Islas-Villanueva *et al.*, 2012).

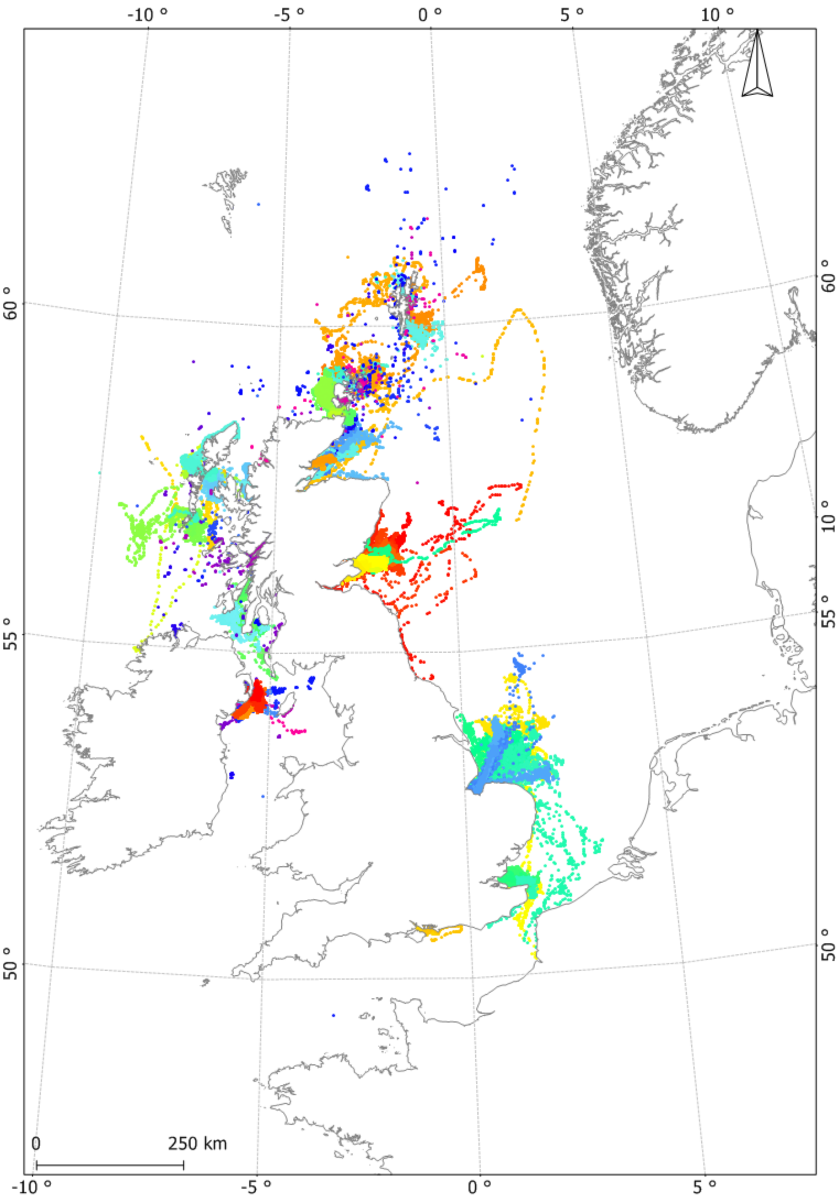


Figure 4.3. Tracks of individual harbour seals tagged at a variety of different locations round the UK. (map created by D. Russell, SMRU). All deployments used to generate these maps are described in Russell *et al.* (2011a). The tags transmit non regular seal location data with the duration of data varying between individual deployments. All locations have been cleaned according to SMRU protocol (Russell *et al.*, 2011b). Where appropriate telemetry locations were then corrected for positional error using a linear Gaussian state–space Kalman filter (Jones *et al.*, 2011).

Following increasing understanding of harbour seal populations, WGMME reiterates its recommendation to OSPAR that the EcoQO subunits be updated. The revised harbour seal EcoQO should therefore read:

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 twelve subunits of the North Sea. These subunits are: Shetland; Orkney and north coast of Scotland; Moray Firth and east coast of Scotland; the Greater Wash/Scroby Sands; the French North Sea *and Channel coasts; the Netherlands Delta area; the Wadden Sea; Heligoland; Limfjord; the Kattegat; the Skagerrak; the Oslofjord; and the west coast of Norway south of 62°N.*

Such a change in the subunits would more accurately reflect current monitoring and/or management areas. For the development of the MSFD indicators is recommended, however, that the subunits do not get specifically listed. Thus, avoiding the need to rewrite/update the wording of the indicator as new information on populations comes to light. Recent genetic analysis by Islas-Villanueva *et al*. (2012) assessed the population structure of harbour seals around the UK. Results supported the management regions defined for harbour seals in Scotland (see ToR A).

4.2.2 Grey seal

The EcoQO for grey seal states:

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. These subunits are: Orkney; Fast Castle/Isle of May; the Farne Islands; Donna Nook; the French North Sea and Channel coasts; the Netherlands coast; the Schleswig-Holstein Wadden Sea; Heligoland; Kjørholmane (Rogaland).

In 2010 OSPAR assessed the EcoQO, based upon five years of data to 2006, and concluded it was met for all significant units of the North Sea population (Figure 4.4).

In 2009, the WGMME noted that the Wadden Sea grey seal EcoQO should be changed, as circumstances make it impossible to meet the proposed requirements to survey pup numbers. It is recommended to use moult counts for this area instead; though the importance to continue efforts in obtaining pup count data was noted, in order to compare with available data from the UK. Changes to the stated subunits were also suggested with the 'Isle of May/Fast Castle' sub-unit now being the 'Firth of Forth colonies' in order to include new colonies in this area and, similarly, for the Donna Nook subunit to be altered to 'Greater Wash'. It was also suggested that the Schleswig-Holstein Wadden Sea should be changed to Wadden Sea. In 2011, a new colony of grey seals was noted at Niedersachsen in the Wadden Sea (Trilateral Seal Expert Group, 2011b).

In 2010 OSPAR assessed the EcoQO, based upon five years of data to 2006, and concluded it was met for all significant units of the North Sea population (Figure 4.4).

It therefore recommended that the EcoQO subunits be further updated by OSPAR. The revised grey seal EcoQO should read:

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. These subunits are: Orkney; Firth of Forth; the Farne Islands; the Greater Wash; the French North Sea and Channel coasts; the Wadden Sea; Heligoland; Kjørholmane (Rogaland).

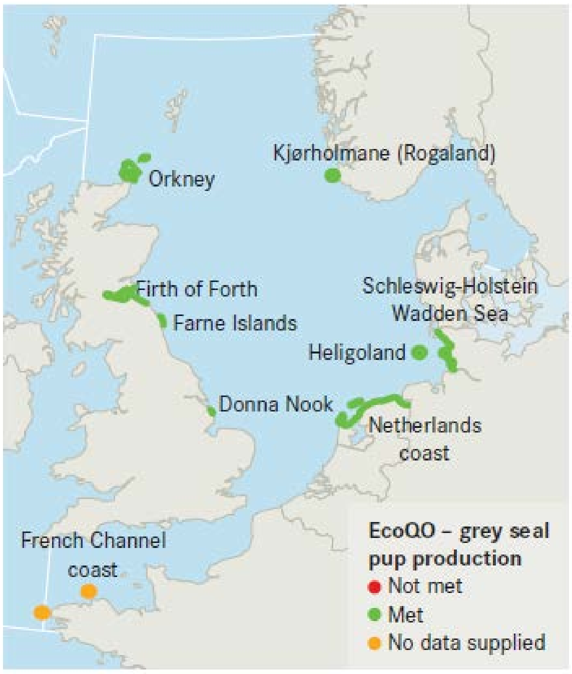


Figure 4.4. 2010 EcoQO assessments for grey seal (from OSPAR, 2010).

Available data (yet unpublished) indicate that orange spots in France would now be green and that other colonies are now present and rapidly increasing in eastern English Channel and southern North Sea (Figure 4.5).

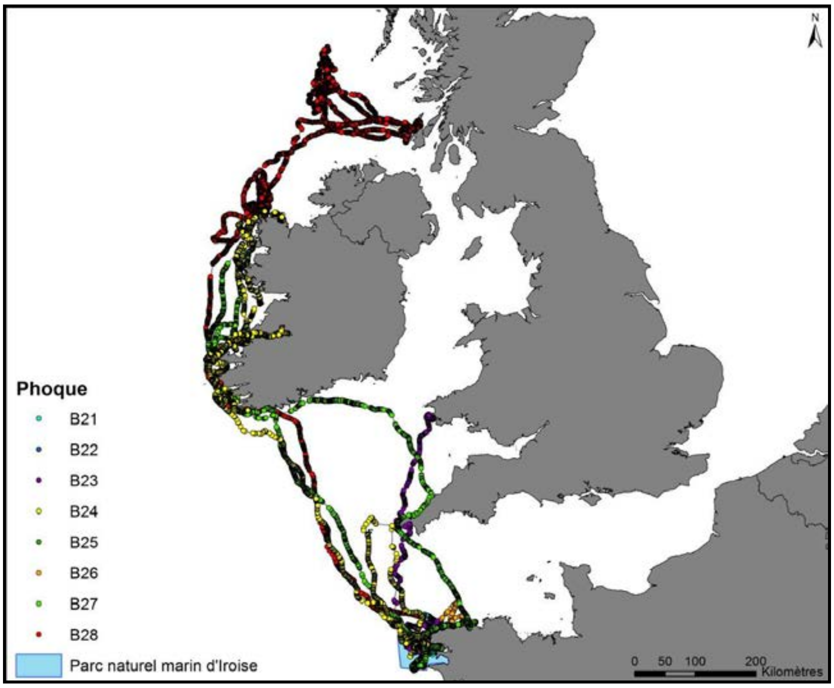


Figure 4.5. Location of breeding sites for grey seals in France and movements of tagged individuals from the area (V. Ridoux, pers. comm.).

4.3 Information needs identified by ICG COBAM for seal indicators

4.3.1 Abundance, distribution and management units for harbour seals

Harbour seals (*Phoca vitulina*) are found around the coasts of the North Atlantic and North Pacific from the subtropics to the Arctic. Five subspecies of harbour seal are recognized. The European subspecies, *Phoca vitulina vitulina*, ranges from northern France in the south, to Iceland in the west, to Svalbard in the north and to the Baltic Sea in the east. The largest population of harbour seals in Europe is in the Wadden Sea which is still increasing after the 2002 PDV epidemic, although the rate of increase is now slowing (Table 1; Trilateral Seal Expert Group 2011a). The populations in the English Channel and the Baltic are also increasing, but elsewhere declines have been recorded in many populations (Table 4.1; Hassani *et al*., 2010).

Harbour seal counts are undertaken during the moult period when greatest numbers are hauled out. Not all individuals are therefore counted during surveys because at any one time a proportion will be at sea. The survey counts are presented as a minimum population estimate. In the UK, satellite tags have been used to track haul out behaviour during the moult and to derive a multiplier to convert counts into total population size (Figure 4.3). Results have indicated that approximately 72% (CI 54% to 88%) of the population will be available to be counted during the normal survey period (SCOS, 2011).

The Wadden Sea management area suffered severe declines of ca 50% during both PDV outbreaks, but abundance is now greater than before the outbreaks, though rates of increase have declined since 2007 (Trilateral Seal Expert Group, 2011a). There is a small population in the Limfjord in northern Denmark, which has not recovered since the PDV outbreak in 2002 (Aarhus University data). In Kattegat, numbers of seals at the Danish haul-outs during the moult have almost doubled since 2002, from ca. 1700 to 3000 (Aarhus University data).

The status of local harbour seal populations (Figure 4.6) varies around the UK. However, the 2010 surveys indicated that there has been either no change or in some cases a large increase in numbers over the previous counts. The latest survey results confirm that since the late 1990s numbers have declined by some 35–85% depending on the haul out site (SCOS, 2011). In contrast, since the 1990s, the Strathclyde population has shown wide fluctuations but recent surveys indicate little overall change since this period. In the Wash, the 2010 count was similar to the pre-PDV count in 2001.

Table 4.1. Size and status of European harbour seal populations. Data are counts of seals hauled out during the moult (taken from SCOS, 2011).

Region	Number of seals counted[1]	Years when latest information was obtained	Possible population trend[2]
Outer Hebrides	1,800	2008	Declining
Scottish W coast	11,400	2007-2009	None detected
Scottish E & N coast	1,600	2010	Declining
Shetland	3,000	2009	Declining
Orkney	2,700	2010	Declining
Scotland	**20,400**		
England	**4,200**	2008	Recent decline[3]
Northern Ireland	**1,200**	2002	Decrease since '70s
UK	**25,900**		
Ireland	2,900	2003	Unknown
Wadden Sea-Germany	10,200	2010	Increasing after 2002 epidemic
Wadden Sea-NL	5,000	2010	Increasing after 2002 epidemic
Wadden Sea-Denmark	2,800	2010	Increasing after 2002 epidemic
Lijmfjorden-Denmark	1,050	2008	Recent decline [3]
Kattegat/Skagerrak	11,700	2007	Recent decline[3]
West Baltic	750	2008	Increasing
East Baltic	600	2008	Increasing
Norway	6,700	2006	Declining
Iceland	12,000	2006	Declining
Barents Sea	700	2008	Unknown
Europe excluding UK	**54,400**		
Total	**80,300**		

[1] –counts rounded to the nearest 100. They are minimum estimates of population size as they do not account for proportion at sea and in many cases are amalgamations of several surveys.
[2] – There is a high level of uncertainty attached to estimates of trends in most cases.
3 – Declined as a result of the 2002 PDV epidemic.
Data sources: www.smru.st-and.ac.uk; ICES Report of the Working Group on Marine Mammal Ecology 2004; Desportes,G., Bjorge,A., Aqqalu, R-A and Waring,G.T. (2010) Harbour seals in the North Atlantic and the Baltic. NAMMCO Scientific publications Volume 8.
Nilssen K, 2011. Seals – Grey and harbour seals. in Agnalt A-L, Fossum P, Hauge M, Mangor-Jensen A, Ottersen G, Røttingen I,Sundet JH, & Sunnset BH. (eds). Havforskningsrapporten 2011. Fisken og havet, 2011(1).;
Härkönen,H. & Isakson,E. 2010. Status of the harbor seal (Phoca vitulina) in the Baltic Proper. NAMMCO Sci Pub 8:71-76.;
Olsen MT, Andersen SM, Teilmann J, Dietz R, Edren SMC, Linnet A,. & Härkönen T. 2010. Status of the harbour seal (Phoca vitulina) in Southern Scandinavia. NAMMCO Sci Publ 8: 77-94.

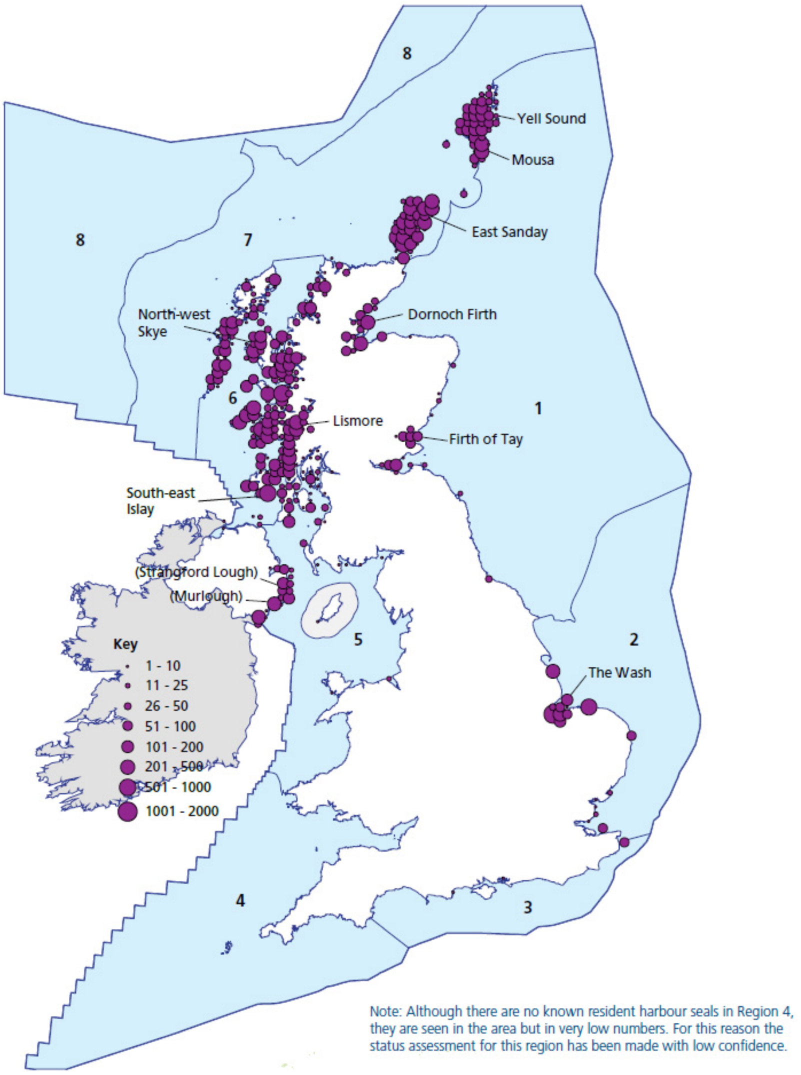


Figure 4.6. The distribution and number of harbour seals in Great Britain and Northern Ireland in August, by 10 km squares, from surveys carried out between 2000 and 2006. Text labels identify the Special Areas of Conservation (SACs) where harbour seals are one of the main reasons for the creation of the protected site. Site names in brackets are SACs where harbour seals have only contributed to the reasons for designation and are not the main reason for the creation of a SAC.

In Ireland, haul-out sites occurred along the entire coastline, with discernibly lower numbers of harbour seals recorded along the southern and eastern seaboards (Figure 4.7; Cronin, 2011). Management areas have yet to be decided upon in Ireland. As elsewhere in Europe, Irish stocks of harbour seals were affected by outbreaks of phocine distemper virus (PDV) in 1988–1989 and 2002 but in the absence of reliable population data at that time meant the impacts of the disease could only be suggested (Cronin *et al.*, 2007). Harbour seal population monitoring in Northern Ireland indicates a consistent decline in the breeding population, with a minimum population estimate of 1248 in 2002 (Duck, 2006) setting an effective baseline for the region. However, with little known about the population inhabiting the rest of the island, these findings have been difficult to place into a wider context (Cronin *et al.*, 2007).

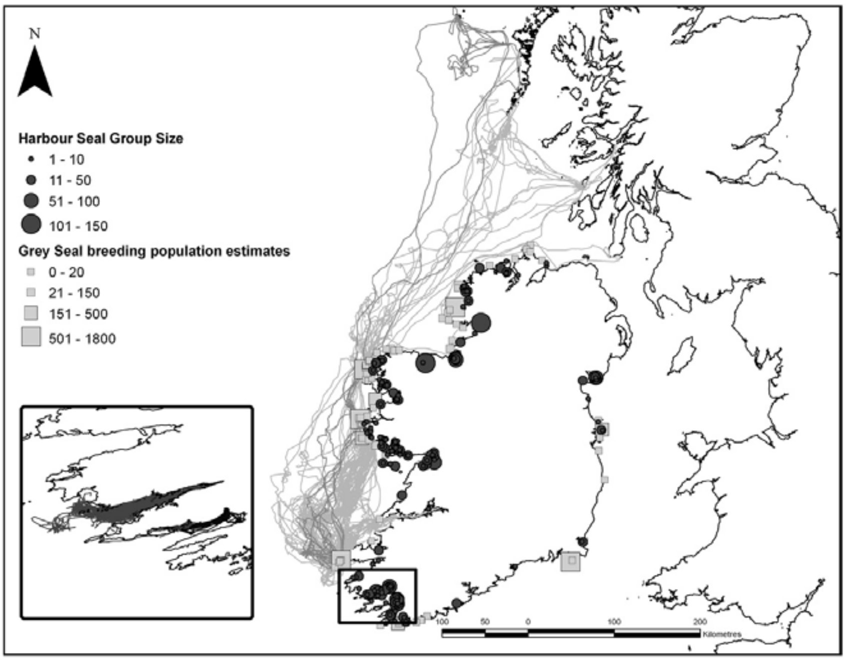


Figure 4.7. Location of haul out and breeding sites for harbour and grey seals, respectively, in the Republic of Ireland including tracks of tagged grey seals (taken from Cronin, 2011).

The counts made at the site level are combined to provide management area estimates within the national level, whilst management considerations need to be made at the international level taking movement between management areas into account. It is recommended that as the MSFD indicators for the harbour seal are further developed that this is approach is borne in mind.

4.3.1.1 Abundance, distribution and management units for grey seal

There are two centres of grey seal abundance in the North Atlantic; one in Canada and the northeast USA, centred on Nova Scotia and the Gulf of St Lawrence and the other around the coast of the UK especially in Scottish coastal waters (SCOS, 2010). Other smaller populations exist in Ireland, Norway, Iceland, the Faroe Islands, Russia, the southern North Sea/English Channel and the Baltic Sea. Populations in Canada, USA, UK and the Baltic are increasing, although numbers are still relatively low in the Baltic where the population was drastically reduced by human exploitation and reproductive failure probably due to pollution.

Grey seal population estimates are most often based on pup counts. The variation in the number of pups born is used as an indicator of population size and, with sufficient understanding of population dynamics, enables the estimation of the total number of grey seals. In the UK, annual aerial surveys are used to determine the number of pups born and a population model is applied to the estimates of pup production to estimate the population size (SCOS, 2011). In other areas, however, it has been found that these pup counts are not as accurate for determining population size as surveys undertaken during the moulting period. Therefore, it is trend in pup counts that is compared between different nations to provide an indication of trends in the European population (Table 4.2).

Table 4.2. Relative sizes of grey seal populations. Pup production estimates are used because of the uncertainity of overall population size (taken from SCOS, 2011).

Region	Pup Production	Years when latest information was obtained	Possible population trend[2]
UK	50,200	2010	Increasing
Ireland	1,600	2005	Unknown[1]
Wadden Sea	400	2008	Increasing [2]
Norway	1,300	2008	Unknown[3]
Russia	800	1994	Unknown[2]
Iceland	1,200	2002	Declining[2]
Baltic	4,700	2007	Increasing[2,5]
Europe excluding UK	**10,000**		**Increasing**
Canada - Sable Island	62,000	2008	Increasing[4]
Canada - Gulf St Lawrence + Eastern Shore	14,400	2007	Declining[6]
USA	2,600	2008	Increasing[7]
WORLD TOTAL	**137,700**		**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] Nilssen K, 2011. Seals – Grey and harbour seals. in Agnalt A-L, Fossum P, Hauge M, Mangor-Jensen A, Ottersen G, Røttingen I,Sundet JH, & Sunnset BH. (eds) 2011.Havforskningsrapporten 2011. *Fisken og havet*, 2011(1).
[4] 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
[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] Thomas,L.,Hammill,M.O. & Bowen,W.D. 2007 Estimated size of the Northwest Atlantic grey seal population 1977-2007 Canadian Science Advisory Secretariat: Research Document 2007/082 pp31.
[7] NOAA (2009) http://www.nefsc.noaa.gov/publications/tm/tm219/184_GRSE.pdf

4.3.1.1.1 Grey seal subunits in the OSPAR area

The subunits for grey seals in the North Sea have already been defined for the EcoQO. These subunits are: Orkney; Firth of Forth; the Farne Islands; the Greater Wash; the French North Sea and Channel coasts; the Wadden Sea; Heligoland; Kjørholmane (Rogaland).

In the Wadden Sea of the Netherlands, Germany and Denmark, the grey seals are regarded as one management area. A maximum of 3000 grey seals have been counted during the moult, while around 500 pups have been born annually in recent years. Numbers of grey seals have increased by 15.5% annually since 2007, which can be explained by immigration from the UK in addition to pup production in the area (Trilateral Seal Expert Group 2011b).

Out with the North Sea, the haul-out locations are well known in the UK (Figure 4.8) which are grouped into defined management areas in Scotland. These are equivalent to the EcoQO subunits. UK grey seal pup production in 2010 was estimated to be 50,174 with pup production remaining stable in the Inner and Outer Hebrides but increasing elsewhere, with the total UK grey seal population at the start of the 2010 breeding season estimated to have been 111 300 (95% CI 90 100–137 700) (SCOS, 2011). Tagging of grey seals from various colonies has been occurring in UK colonies for a number of years (Figure 4.9). Currently this data is being analysed to identify important areas away from the coast for the species.

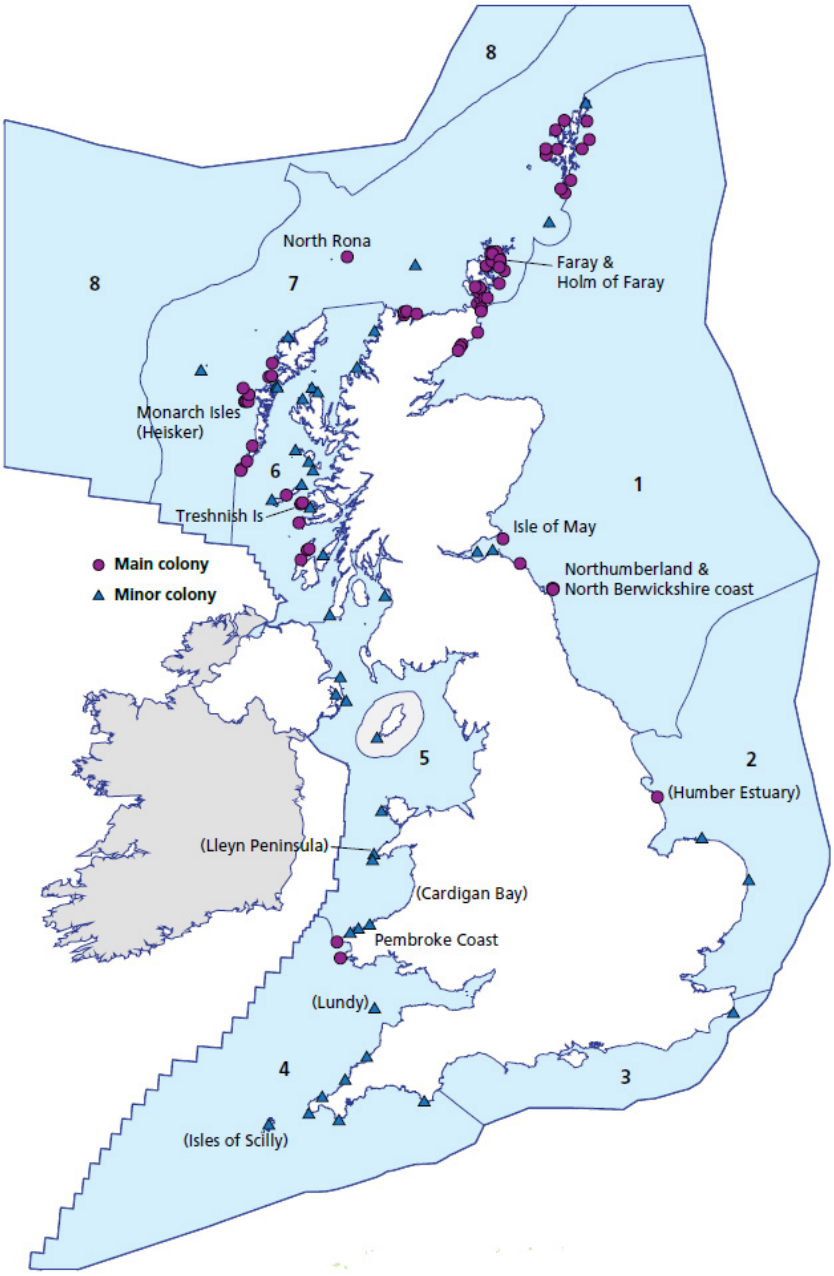


Figure 4.8. The location of grey seal breeding colonies in Great Britain and Northern Ireland. Text labels identify the Special Areas of Conservation (SACs) where grey seals are one of the main reasons for the creation of the protected site. Site names in brackets are SACs where grey seals have only contributed to the reasons for designation and are not the main reason for the creation of a SAC.

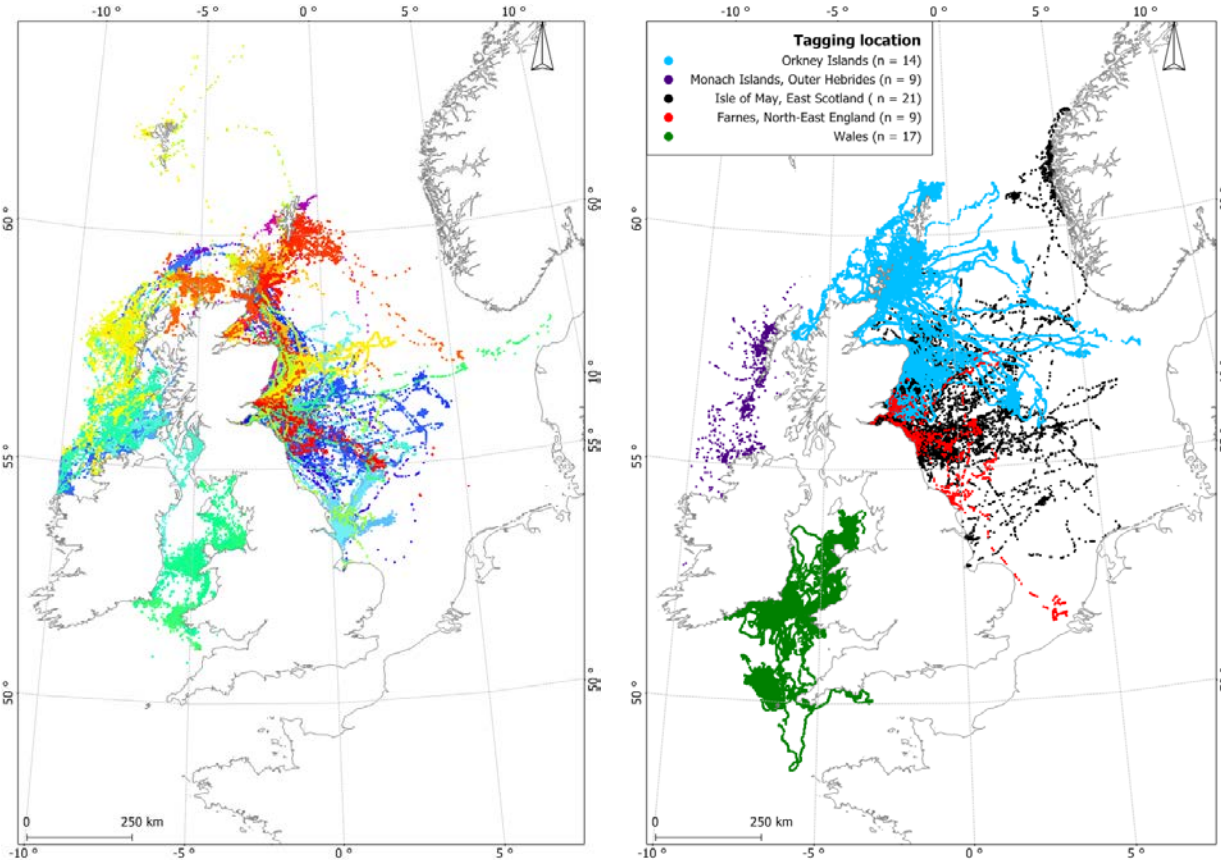


Figure 4.9. Tracks of individual grey seals >1year (left) and pup tracks by breeding colony (right) tagged at a variety of different locations round the UK (maps created by D. Russell, SMRU). All deployments used to generate these maps are described in Russell *et al.* (2011a). The tags transmit non regular seal location data with the duration of data varying between individual deployments. All locations have been cleaned according to SMRU protocol (Russell *et al.*, 2011b). Where appropriate telemetry locations were then corrected for positional error using a linear Gaussian state–space Kalman filter (Jones *et al.*, 2011).

In Ireland, over 80% of the population is associated with seven key breeding locations along the east, southeast and Atlantic coasts (Figure 4.6; Cronin, 2011). Currently, pup production is being surveyed and it is expected that an update will become available in 2013 for comparison to the 2005 baseline estimate (M. Cronin, pers. comm.). Telemetry studies have demonstrated that grey seals from SW Ireland regularly move up the Irish west coast and north to the Scottish west coast (Figure 4.7; Cronin, 2011). Management areas have yet to be defined in Ireland, but currently the region is treated as a single unit in European estimates for the population (e.g. Table 4.2).

Population estimates are made for each of these management areas (or EcoQO subunits) separately. It should, however, be noted that there is evidence of movement between these management areas from both photo ID and telemetry studies. So, although population estimates are made at the site level and combined to provide management area level estimates, management considerations need to be made at the international level taking movement between management units into account.

4.3.2 Monitoring and baselines for seal indicators

Monitoring of seals is site based. For grey seals this is largely based on breeding colonies and for harbour seals on haul-out sites. 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. The current scheme of 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 current harbour seal monitoring programme in the Wadden Sea had sufficient power (80%) to detect a minimal trend in ten and six years of respectively approximately 2.2 % and 6% per annum as long as the variance within years is stable around the mean value. If the within-year variance increased, the power to detect trends may decrease rapidly.

A similar 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.

The seal EcoQOs reflect the general status of species and many factors could underlie any declines noted. Currently, where the EcoQO is not met, this should trigger research by the relevant nation to identify the cause of the changes observed. Where problems result from human activities, suitable management measures should then be put in place. Such an approach seems aligned with the intended aims of the MSFD. Additionally, HELCOM (2012b) proposed that GES should be considered as being met if the decrease of the population is less than 10% over a period of ten years (i.e. similar to the OSPAR EcoQO for seals) although no power analyses on the effectiveness of the existing survey schemes appear to have been conducted. In contrast to the OSPAR and HELCOM approaches, ICG-COBAM has proposed a more generic approach: 'No statistically significant decrease with regard to baseline due to anthropogenic activities'.

ICG – COBAM recommended that the baseline against which GES is set should be historical, i.e. prior to human influence. Considering that the ecosystems have changed fundamentally over the last 100 years, it is not possible to identify an historical baseline that is 'pre human influence' for determining 'good environmental status'. Additionally, it should be noted that there are no accurate estimates of seal population levels prior to the start of current monitoring programmes. The best option for a population estimate baseline would be the 'favourable reference population' as defined under the Habitats Directive's Favourable Conservation Status assessments.

WGMME **recommends** that power analyses are undertaken for all seal management areas to determine the trends in populations that can accurately be assessed with current monitoring practices. It is also recommended that ICG-COBAM give consideration to using the EcoQO approach rather than the more generic baseline proposed but that it would be useful to relate the percentage change to some earlier baseline such as the favourable reference population determined under the Habitats Directive.

4.4 Development of cetacean indicators

A variety of Member States all proposed similar indicators for cetaceans. As with the proposed indicators for seals, these were largely based on monitoring already required for other European Legislation (e.g. Habitats Directive and EU Regulation 812/2004). These indicators included:

1) **Criterion:** Species distribution. **Indicator:** Distributional range and distributional pattern with range. **Parameter/metric:** Distributional range and pattern of cetaceans species regularly present;
2) **Criterion:** Population size. **Indicator:** Population abundance and/or biomass, as appropriate. **Parameter/metric:** Abundance at the relevant temporal scale of cetacean species regularly present;
3) **Criterion:** Population condition. **Indicator:** Mortality rate. **Parameter/metric:** Numbers of individuals within species being bycaught in relation to population.

The information requirements associated with these were:

1) Compilation of existing data on abundance. This has already begun through the Joint Cetacean Protocol (JCP), but it is recognised that a number of significant national datasets are missing.
2) Development of a baseline for each species.
3) Development of a standardized monitoring methodology, or alternatively a mechanism for standardizing data post collection. Effort-related monitoring of cetaceans is to some extent standardised. It is the standardising of data post collection that is necessary, although this has already been started through the JCP.
4) Development of an assessment tool.

Bycatch will be dealt with in Section 4.5. The OSPAR meeting (2011) also noted that very few indicators had been proposed by Member States that utilised the generally well developed datasets collected through strandings schemes. One other aspect of cetacean work that has not been considered is that of biopsy sampling. It was felt that the potential to develop indicators from both these areas of work should be further explored as part of this ToR.

HELCOM (2012b) proposed an indicator based on the historical distribution of harbour porpoises in the Baltic Sea with the parameter: presence as indicated by the frequency of registrations per area in a year (e.g., >ten registrations/1000 km^2). '*The current population size of the Baltic harbour porpoise is extremely small and due to its low abundance no longer reliably quantifiable. Therefore, it appears impracticable to propose the abundance of harbour porpoise as a state indicator or a quantitative target for abundance as a conservation goal. At extremely low densities, such target would be almost impossible and very costly to monitor. The Baltic porpoise population has not only dwindled in numbers to less than 250 reproducing adults (IUCN 2008) but also evacuated large parts of its historic range throughout the Baltic Proper. Therefore, the extent of the distribution range appears to be a suitable proxy for population size assuming that an increasing population would also be likely to expand its range. Anecdotal information on (pre-industrial) porpoise distribution indicates a probably continuous distribution throughout the Baltic Proper, possibly also covering the entire Gulf of Bothnia as well. Therefore, a regular basin-wide presence could serve as proxy for successful population recovery*.' Further details of the HELCOM bycatch indicator for marine mammals are given in Section 4.5.

4.4.1 Monitoring cetaceans

Ideally a monitoring strategy for marine mammals should inform on trends and status of marine mammal populations at spatio-temporal scales and resolutions that are consistent with the size of the conservation units and the dynamics of the pressures that marine mammal populations are exposed to. Because MSFD general goals are to recover GES across EU waters, it can be inferred that when it comes to marine mammal populations the objective would be to restore or maintain population in a healthy state. Irrespective of how healthy state is defined, this would imply that the monitoring strategy should be able to detect trends, in particular negative ones, before they have resulted in measurable changes in absolute abundance. This is particularly an issue for cetaceans as they are notoriously difficult and expensive to census with accuracy at short time intervals. In this context a monitoring strategy should combine decadal standardized surveys aiming at estimating absolute abundance with other approaches that would continuously inform on trends in relative abundance, encounter rates, occurrence, distribution, or any feature that is on the causal chain that links pressure to abundance (Figure 4.10).

Pressures are diverse and act at various levels. For clarification, one can propose a typology of pressures. Primary pressures would be those removing individuals from the population by adding man-induced mortality to the existing natural mortality. Bycatch, collision, hunting, deliberate killing and some acute sound exposures are examples of primary pressures. Secondary pressures would act by degrading condition and health and, as a consequence, vital rates. Contaminants and depleted food resources are such secondary pressures that can affect overall body condition and more specifically fertility and immune function, leading to lower demographic performances. Finally, tertiary pressures would be those acting by displacing populations toward habitats of poorer quality or precluding the proper accomplishment of vital functions. Disturbance in critical habitat, ambient noise, habitat modification can generate such behavioral dysfunction. The impact of primary pressures can be assessed provided that estimates of removal and abundance are available. Assessing the impact of secondary pressures on populations is at its beginning and assessing the impact of tertiary pressures on populations is not possible yet.

MONITORING: 1	2	3-4-5	3-4	3-4-6-7

Figure 4.10. Schematic representation of the causal link between pressure and marine mammal status. Pressure monitoring is provided by human activity surveillance systems (1). Monitoring the risk is typically the goal of impact assessment programs (2); another angle of view is the identification spatial overlap between marine mammal and human activity distributions. Monitoring condition and health generally implies access to whole carcasses or to a suite of biological samples, most often provided by stranding schemes (3). Alternatively, photographic-based approaches can be useful (4). Direct observation and telemetry can document behavioral responses to pressures (5). Demographic features or vital rates can be obtained from age and reproductive status obtained transversally from stranding (3) or longitudinally from photo-ID (4). Abundance and distribution are ideally documented by dedicated surveys (7), but information from stranding data (3), photographic approaches (4; can also document abundance by CMR analyses for some small populations) and platforms of opportunity (6) can usefully contribute information about relative trends.

4.4.2 Compilation of existing data on abundance

WGMME 2008 (Section 8.4.5) discussed the issue of power to detect trends in abundance, survey design and over vs. under protection decisions that derive from the choice of β and α levels: '*The statistical power of a monitoring program is the probability that the monitoring will detect a trend in the data despite the 'noise' associated with seasonal cycles and other fluctuations (Nichols and Williams, 2006)….if the risk of over and under-protection are to be similar, a trade-off is required between power (i.e. β) and level of significance (i.e. α) with consideration given to using a value of 0.2.*'

WGMME 2009 (see Section 9.3.2) considered these issues further and looked at the power of the large-scale SCANS and CODA surveys to detect trends in abundance over time. '*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.*'

As part of improvements in approach to assessments, a collaborative project, the Joint Cetacean Protocol (JCP) has been developed, which will deliver information on the distribution, abundance and population trends of cetacean species occurring in NW European waters. It is intended that the project outputs will assist governmental reporting to various Directives (e.g. The Habitats Directive and the Marine Strategy Framework Directive) and will also improve the robustness of marine Environmental Impact Assessments. This initiative has been welcomed internationally by ASCOBANS as a mechanism for improving small cetacean assessments (ASCOBANS, 2009).

The JCP brings together effort-related cetacean sightings data from a variety of sources including large scale international surveys such as SCANS I & II and CODA, surveys based on platforms of opportunity such as ICES International Bottom Trawl Surveys (European Seabirds at Sea (ESAS) cetacean data), as well as more localised non-governmental data (e.g. SeaWatch Foundation and ARC) and industry data (e.g. that collected in relation to potential renewable energy installations). These data, collected between 1979 and 2010, represent the largest NW European cetacean sightings resource ever collated and have been standardised to a common format, checked and cleaned. It should be noted that the JCP is heavily dominated by UK lead survey work. Other sources should be encouraged to join JCP in the future, notably from waters other than UK similarly collected from dedicated surveys or platforms of opportunity.

For harbour porpoises, bottlenose dolphins and common dolphins in the Irish Sea, Paxton and Thomas (2010) reported that quite small declines in modelled population density (0.3–2.2% per year) over a 6-year reporting period could be detected with power of 0.8, for the latter part of the survey period. For other species and earlier time periods, only very large changes in modelled population density would be detectable. However, the modelled population densities rely on spatial and temporal smoothing, and hence sudden declines would not necessarily be detectable.

The models developed by Paxton and Thomas (2010) have been further refined and expanded to include the Scottish west coast (Paxton *et al.*, 2011). Density surfaces varying in time were generated for harbour porpoise, minke whale, bottlenose dolphin, short-beaked common dolphin and white-beaked dolphins; with a non-temporal model used for Risso's dolphin. The density surfaces proved complex to model and some bootstrap confidence intervals were very wide especially in areas of low effort and associated with high predictions.

For harbour porpoises, monthly abundances were found to peak in August and there is evidence for a strong temporal trend. Estimated numbers fluctuated in their high season (summer) between 10 200 (CI: 5500–17 700, CV: 0.30) in 1991 and 107 900 (CI: 87 800–142 000, CV: 0.13) in 2005. A predicted model surface for August 2010 is given in Figure 4.11 with 2.5% and 97.5% confidence limits for each cell illustrated in Figure 4.12.

The outputs of the JCP project covering the European North Atlantic area (Figure 4.13) are expected later in 2012 and will include:

- Annual estimates of species-specific cetacean abundance (with 95% confidence intervals) at a Regional Seas scale, suitable for Habitats Directive and MSFD reporting.
- Species-specific summary datasets depicting cetacean distribution and relative abundance at a range of resolutions with advice on the most robust resolution. Where there is sufficient data, density surface plots will be produced for each season annually, with an assessment of trends over time and the power to detect these trends. It is expected that the power to detect trends over this area are unlikely to be as high as those reported for the Irish Sea subset in Paxton and Thomas (2010).

Currently the area covered by the JCP modelling does not match with the requirements for transboundary reporting area under the Habitats Directive for the Marine North Atlantic region. There is a considerable lack of data for the Bay of Biscay and further south along the Spanish and Portuguese coasts which has prevented the surface density modelling from extending into this region. Surveys in these areas have either been ongoing for a number of years or have been initiated in recent years by France, Spain and Portugal. If combined with the SCANS II data for the region, inclusion of this data would significantly improve the modelling capability into this area. It is also recognised that there are significant datasets from the North Sea area missing from the JCP dataset, notably the annual aerial surveys undertaken in German and Dutch waters. It is recommended that future developments of the JCP result in it becoming a tool suitable for transboundary assessments such as those for favourable conservation status (Habitats Directive) and good environmental status (MSFD).

It is **recommended** that WGMME assess the JCP outputs when they become available with a view to their contribution to international reporting requirements in 2013. The current Article 17 guidance for the 2013 reporting round includes a much greater emphasis on transboundary reporting where appropriate (European Commission, 2011). Further development/refinement of MSFD indicators of biodiversity will be required through 2013 and implementation of the monitoring needs to meet these requirements is needed by 2014. Development of an international equivalent of the JCP is also recommended that could be held by ICES.

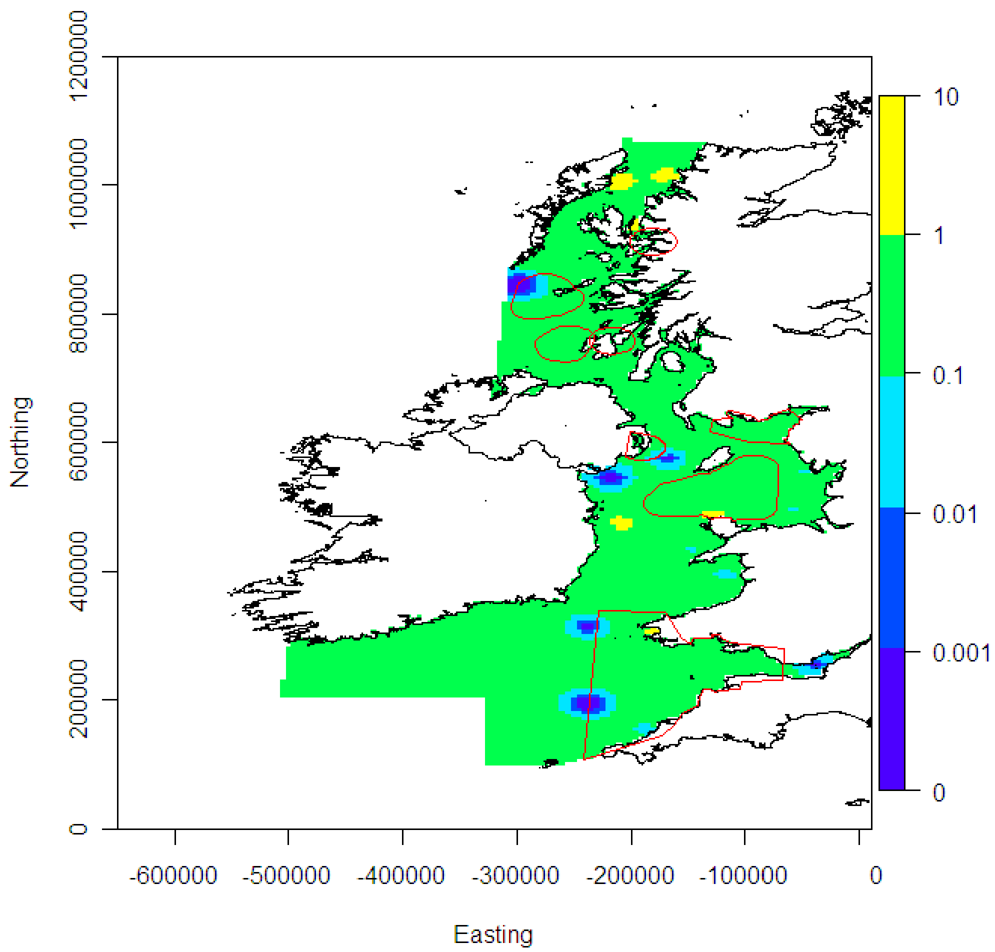
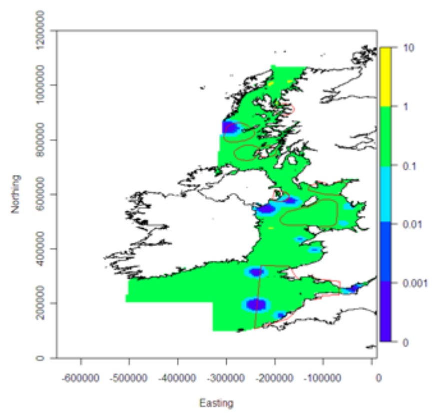


Figure 4.11. Predictions for harbour porpoise density for the August 2010. Colours indicate densities (animals/km²). N.B. Colour breaks are on a log scale. The red boxes indicate industry search areas. (From Paxton *et al.*, 2011)

a.



b.

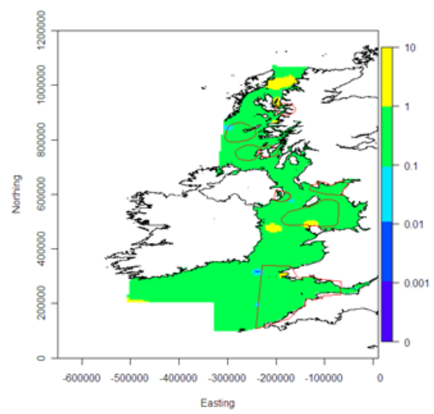


Figure 4.12. Uncertainty in the per cell density estimates for harbour porpoise August 2011, a. the 2.5% confidence limit on each prediction grid cell. b. 97.5% confidence limit on each prediction grid cell. Colours indicate densities (animals/km²). N.B. Colour breaks are on a log scale.

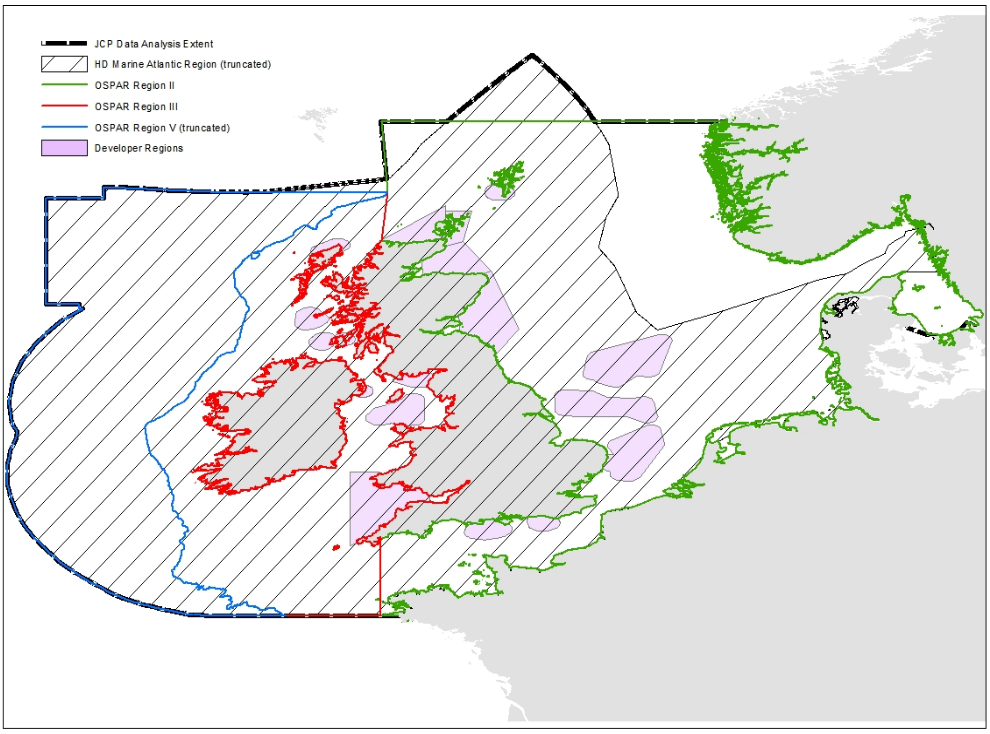


Figure 4.13. Current extent of planned JCP output and regions of interest.

4.4.3 Development of a baseline for each species

ICG – COBAM recommended that the baseline against which GES is set should be historical, i.e. prior to human influence and pre-industrial whaling for relevant species. Considering that the ecosystems have changed fundamentally over the last 100 years, it is not possible to identify an historical baseline that is 'pre human influence' for determining 'good environmental status' for cetaceans. Although techniques have evolved that enable the determination of historic population size through genetic analyses of modern and museum specimens (e.g. Alter *et al.*, 2007; Wandeler *et al.*, 2007), such approaches are only applicable to species that were commercially hunted. It is unlikely that there is suitable sample availability for species for which were not commercially hunted (e.g. the majority of small cetacean species) in European waters. Consequently, there are few accurate estimates of cetacean population levels prior to the start of modern monitoring programmes.

In contrast, HELCOM (2012b) were able to propose an indicator taking such an approach because the 'current population size of the Baltic harbour porpoise is extremely small and due to its low abundance no longer reliably quantifiable....The Baltic porpoise population has not only dwindled in numbers to less than 250 reproducing adults (IUCN, 2008) but also evacuated large parts of its historic range throughout the Baltic Proper. Therefore, the extent of the distribution range appears to be a suitable proxy for population size assuming that an increasing population would also be likely to expand its range. Anecdotal information on (pre-industrial) porpoise distribution indicates a probably continuous distribution throughout the Baltic Proper, possibly also covering the entire Gulf of Bothnia as well. Therefore, a regular basin-wide presence could serve as proxy for successful population recovery.'

Once the JCP output becomes available in 2012, it is expected that the most appropriate baseline for each species (e.g. favourable reference condition for FCS and baselines for MSFD indicators for GES) could be determined, and the power to detect trends from this baseline elucidated.

4.5 Bycatch indicators

Bycatch was identified by most Member States as being the most significant anthropogenic impact on marine mammals, and cetaceans in particular. There are a number of policy drivers that have lead to a focus on bycatch as a potentially indicator of GES for the MSFD. These are the Council Regulation 812/2004 and the Data Collection Regulation, as well as Article 12 of the Habitats Directive (concerning incidental killing and capture) and OSPAR's EcoQO for harbour porpoise bycatch in the North Sea. The ICG-COBAM indicator sheets summarised the indicator as:

- **Criterion:** Population condition. **Indicator:** Mortality rates. **Parameter/metric:** Numbers of individuals within species being bycaught in relation to population. **Target:** the annual bycatch rate of [marine mammal species] is reduced to less than [X]% of the best population estimate.

ICG-COBAM identified the following aspects of the proposed indicator that needed further development:

1) Target (in %) for other species than the harbour porpoise.
2) Population (or management unit) against which to set the target (developed already for harbour porpoises in the North Sea), and development of a baseline for each population or management unit.
3) Development of a standardized monitoring methodology for bycatch or alternatively a mechanism for standardizing data post collection. This is currently being progressed through WGBYC.
4) Development of an assessment tool. This is currently being progressed through WGBYC.

HELCOM (2012b) also proposed bycatch indicators for marine mammals and stated that the impact should be reduced close to zero. HELCOM (2012b) states that '*for healthy mammal populations (with an abundance ≥80% of a population at carrying capacity) a tolerable bycatch rate may amount to 1.0% (plus another 0.7% anthropogenic take due to other impacts such as pollution, noise etc.) of the local population. For depleted populations such as the "critically endangered" (according to Hammond et al., 2008) porpoise population of the Baltic Proper and the rapidly decreasing (according to Teilmann et al., 2011) porpoise population of the Belt Sea, bycatch was recommended to be reduced to near zero immediately*'. For less critically endangered species, a similar approach is unlikely to be acceptable. Although the ASCOBANS (2006) resolution on incidental takes states that 'anthropogenic removal is reduced by the Parties to below the threshold of "unacceptable interactions" *with the precautionary objective to reduce bycatch to less than 1% of the best available abundance estimate and the general aim to minimise bycatch (i.e. to ultimately reduce to zero).*'

4.5.1 Target (in %) for other species than the harbour porpoise

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). Harbour porpoise populations were modelled under various scenarios of bycatch and target population size using a very simple deterministic population dynamics model with assumed maximum rate of increase of 4%. It applies to a "biological" population with independent population dynamics. The

1.7% figure is the rate of total removals from a population that would still allow the harbour porpoise population to achieve 80% of its carrying capacity over a very long time horizon (a proxy for a sustainable population). The figure has subsequently been adopted by ASCOBANS as the rate above which bycatch would become "unacceptable"; noted by a North Sea Ministerial meeting, developed into an EcoQO by OSPAR and accepted by the European Commission (Anon., 2010) as the level above which ICES might advise that mitigation measures would become necessary. The OSPAR EcoQO on porpoise bycatch in the North Sea states that "*Annual bycatch levels should be reduced to be below 1.7% of the best population estimate*".

Notwithstanding that the figure of 1.7% is a gross over-simplification (as described in IWC 2000), if this management target is to be applied to management units for harbour porpoise, the animals living in the areas defined by these MUs 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.

WGMME reiterates the need for management targets to be determined in the context of explicit conservation and management objectives. It draws attention to the conclusions of its discussions on the use of the management procedure approach to determine safe limits to small cetacean bycatch in the context of specified conservation objectives (WGMME 2008, Section 4 and WGMME 2009, Section 7.2), including the strong recommendation that the bycatch management procedures developed under the SCANS-II and CODA projects (SCANS-II 2008; Winship, 2009; CODA, 2009) should be taken into consideration by DG MARE when reviewing EU Regulation 812/2004 (WGMME 2009, Section 7.3). WGMME (2010) recommended that ICES be encouraged to "move away from implicit and automated conservation targets and towards the explicit definition and justification of target population sizes and management objectives". Subsequently, ICES advice to the European Commission in 2010 (Section 1.5.1.2) included the following recommendations:

'(i) ICES could provide advice to the Commission on the various approaches to establishing specific conservation and management objectives to manage the impacts of fisheries on marine mammal and seabird populations.

(iii) 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.'

One of the targets of the ASCOBANS' North Sea Conservation Plan for Harbour Porpoises is 'to finalise a population dynamics modelling framework for evaluating the effect of bycatches (and other anthropogenic activities) on harbour porpoises in the North Sea that anthropogenic activities do not prevent agreed conservation goals being met.... building upon the advances made by the IWC/ASCOBANS working group, the ICES/SGBYC and the SCANS II project and the recommendations therein and other Actions (2, 3, 4, 7) of this plan including: agreement of operational management objectives by policymakers; finalisation and scientific implementation of a management procedure by scientists; agreement by policymakers to develop and implement management advice based on the results of the management procedure' (ASCOBANS, 2009). Similarly, the ASCOBANS Working group on Bycatch has also indicated that 'the IWC/ASCOBANS working group had recommended a management procedure approach using simulation studies to develop algorithms for setting limits to achieve management objectives' (ASCOBANS, 2012).

These and other issues relating to management rules for marine mammal exploitation have recently been discussed in the literature (Lonergan, 2011; Cooke *et al.*, 2012; Lonergan, 2012). The processes of management strategy evaluation and the management procedure approach to fisheries management, pioneered by the IWC in the development of the Revised Management Procedure (RMP), which aims to maintain cetacean populations above a fixed proportion (72%) of their carrying capacity, and used in the development of the bycatch management procedures described above, are the topic of a session at the World Fisheries Congress in Edinburgh in May 2012. The only appropriate way to incorporate the concept of management units that are not biological populations with more or less independent dynamics is within management frameworks such as those developed for setting limits to safe bycatch for harbour porpoise and common dolphin (SCANS-II 2008; Winship, 2009; CODA, 2009). These projects considered two procedures, one based on PBR which used a single, current estimate of absolute population size as input and one based on the RMP (termed the Catch Limit Algorithm or CLA) that used a time-series of estimates of absolute population size and estimates of absolute bycatch as input. Both procedures were tuned to three different potential conservation objectives:

i) Median population at 80% of carrying capacity after 200 years.

ii) A 95% probability that the population would be at or above 80% of carrying capacity after 200 years.

iii) Worst case scenario with biased input data and a 95% probability that the population would be at or above 80% of carrying capacity after 200 years.

Based on analysis of data on harbour porpoise in the North Sea, Winship *et al*. (2006) suggested that despite being more complex, the advantages conferred by the CLA procedure were sufficient for it to be considered as the best option. They also recommended that management objectives should be precisely specified and that the judgement of which tuning to use could be based on an assessment of the available information.

To enable the development of bycatch indicators, WGMME reiterates its **strong recommendation** that the bycatch management procedures developed under the SCANS-II and CODA projects (SCANS-II 2008; Winship, 2009; CODA 2009) should be taken forward to develop management frameworks for marine mammal bycatch at a European level. The development of bycatch indicators for the MSFD should be based on such an approach rather than a direct transfer of the simplistic percentage approach. It is recommended that WGMME and WGBYC collaborate to progress this work during 2013 for harbour porpoises, common dolphins, as well as grey and harbour seals, as part of the MSFD bycatch indicator developments.

4.5.2 Population (or management unit) against which to set the target (developed already for harbour porpoises in the North Sea), development of a baseline for each population or management unit and development of an assessment tool

WGMME has a standing ToR to assess new information on populations and stock structure of marine mammals. Section 3.1 of this report presents the most recent information on proposed management units for the European North Atlantic for harbour porpoise, common dolphin, bottlenose dolphin, white-beaked dolphin, white-sided dolphin and minke whale.

WGMME **strongly recommends** that Member States use the proposed management units for reporting requirements of the Habitats Directive and for the development of indicators and their assessment for the Marine Strategy Framework Directive. In summary, there is a single MU in European North Atlantic for common dolphin (*Delphinus delphis*), white beaked dolphin (*Lagenorhynchus albirostris*), white sided dolphin (*Lagenorhynchus acutus*) and minke whale (*Balaenoptera acutorostrata*). For bottlenose dolphin (*Tursiops truncatus*) there are ten separate units closely associated with the mainly resident inshore populations in the European North Atlantic and a separate MU for the wider ranging mainly offshore animals. For harbour porpoise (*Phocoena phocoena*), MUs are proposed for the Iberian Pennisula, Bay of Biscay, Celtic Sea (including SW Ireland, Irish Sea and Western Channel) and NW Ireland/West Scotland and the North Sea. The MUs for harbour porpoises may need to be revisited as indicators for MSFD become better defined and aligned with ICES rectangles to enable the calculation of more accurate bycatch estimates. For the purposes of MSFD, it maybe that consideration of the species will need occur at the regional seas level (e.g. North Sea).

4.5.3 Development of a standardized monitoring methodology for bycatch or alternatively a mechanism for standardizing data post collection

In addition to the considerations outlined above, and as a result of the policy interest in bycatch, the development of methods to assess its impact has been ongoing for a number of years. For example, WGMME (2008) tried to evaluate progress to date with the harbour porpoise bycatch EcoQO on a North Sea wide basis. It was quickly apparent that many of the fisheries suspected to have the highest bycatch levels (usually static net fisheries from small vessels) were conducted without bycatch observer programmes as these are not a requirement of Council Regulation 812/2004. Subsequently, SG/WGBYC have tried annually to evaluate the impact of fisheries bycatch through the requirements of Council regulation 812/2004.

The assessment of the impacts of fisheries on non-target species is hampered by poor levels of bycatch monitoring, the inadequate response of Member States to meet their obligations under Regulation 812/2004 and the Habitats Directive, and the limited availability of fleet effort data especially for smaller vessels and recreational fisheries (ICES advice to EC, 2011). Consequently, extrapolated estimates of total bycatch are only available for 2009. There were approximately 870 striped dolphins, 1500 common dolphins, ten bottlenose dolphins and 1100 harbour porpoises (WGBYC, 2011). These totals provide only a very patchy overview of total cetacean bycatches within European waters due to low and uneven sampling coverage. As a consequence, it is not currently possible to evaluate whether such an indicator will provide an accurate assessment of GES, but data collation techniques are continually improving and coverage of the relevant fisheries sectors has been increasing. Although it should be noted that concerns have been raised regarding the change in legislation with respect data collection on vessels. SGBYC (2010) reported that '*in July 2009 the Ministry of Fishing had decided to merge all the EU observer requirements (EC regulations relating to, inter alia, deep-sea species, yellow fin tuna and deep setnets, as well as those mandated under the DCF and Reg 812/2004) and national at-sea monitoring projects. The objective of the changes was to improve economic efficiency in the collection of data and to avoid too many observers interacting with the same vessels. Less easy to address are problems arising from very different stratifications needed to address protected species bycatch and discard or biological sampling. Protected species bycatch monitoring is usually accorded less importance in Europe than discard and biological sampling, and integrating the two schemes runs the risk of* *obscuring objectives for the former. One concern raised over the integration of monitoring in France was that detailed data collection on fish catches and discards could make skippers less likely to agree to take an observer. Furthermore, other problems may arise where fleet stratifications devised to monitor discards are used to monitor protected species bycatch rates, making the raising procedure much more difficult.*'

More recently WGBYC (2012) reported that 'one of the fundamental issues in making the assessment of impact is in the processing and marrying of the abundance and bycatch data, both spatially and temporally. Abundance estimates for cetaceans are available within survey blocks and over the entire survey areas; however, the boundaries do not necessarily delineate true biological populations. Additionally, for most cetacean species in the Northeast Atlantic, there is debate about population structure given the scarcity of data and often conflicting results generated by different approaches. Fisheries data in the Northeast Atlantic are collated by ICES regions and subdivisions which have no bearing on cetacean population structure. Therefore, spatial matching of the two types of data needs careful consideration and definition of spatial units for management purposes is more workable and will allow progress to be made toward assessing population level consequences.'

To improve assessments, WGBYC 2012 further developed the approach devised by WKREV812 to estimate bycatch of porpoises in the North Sea. A fundamental decision was to define 'Management Regions' (MR) consisting of multiple ICES subdivisions which also contained porpoise abundance information. The North Sea management region was defined as ICES Area IVabc. This approach was taken because it was an easier process than trying to spatially redefine fisheries data. This uses the three ICES subareas for the North Sea (IVabc) to estimate occurrence of known relevant fisheries, catch rates from UK fisheries monitoring and the corresponding SCANS abundance estimates to come up with a bycatch estimate of about 1700 animals. 1.7% of the population in this area is approximately 3500 animals. In 2013, WGBYC plan to further develop this approach covering additional species and/or areas where sufficient data are available. This methodology represents the most pragmatic approach in terms of the fisheries data/information available, although it may not match whole population distributions.

As recommended above, WGMME and WGBYC should collaborate in 2013 to develop bycatch management procedures for marine mammals at a European level for harbour porpoises, common dolphins, as well as grey and harbour seals, as part of the MSFD bycatch indicator developments. This should be undertaken prior to further development of the 1.7% approach. Development of an MSFD indicator for bycatch will also require the refinement of species MUs that align with ICES rectangles to enable the calculation of more accurate bycatch estimates. For the purposes of MSFD, it maybe that consideration of the species will need to occur at the regional seas level (e.g. North Sea).

4.6 Potential indicators from strandings

Stranding schemes have been developed in most European countries for many decades, offering a unique large scale and long lasting tool for the continuous surveillance of marine mammal populations collecting information of the causes of mortality (natural and anthropogenic), on health status, disease, contaminants, life history and diet. Stranding data have well known advantages and limitations. Limitations of the stranding data mainly relate to the opportunistic nature of the sampling protocol, the biases in the composition of sampled animals and uncertainties about the geographic origins of the animals. Stranded animals are a biased set of the 'at sea' population, with sick animals, coastal animals and species and individuals which remain buoyant when dead occurring in higher proportions. Despite these biases, monitoring stranded animals can be an efficient and cost effective way enhancing our knowledge of species biology and, in particular, they also have a "sentinel" function, to changing conditions and new threats.

Indicators establishing trends in the anthropogenic impact on cetacean populations over time require certain key assumptions are met. For any given surveillance area, animals should undergo standardized and harmonised necropsies which ensure good observer agreement between cases and confidence in the accuracy of the diagnosis. The cause of death indicator is defined as the primary reason the animal became stranded, categorised into class (e.g. trauma, disease and other) and further subcategorised, if appropriate, into specific cause (e.g trauma includes bycatch, vessel strike, bottlenose dolphin attach, cold stunned, etc.). Categorical cause of death can thus be used to calculate absolute and proportional causes of mortality rates, e.g. of bycatch mortality based on proportion of carcases with diagnosed causes of death which are attributable to bycatch.

Changes in the various causes of death are likely to be indicative of changing anthropogenic influences where those causes of death are associated with human activity. It is particularly important to minimize biases arising from differential capacity to determining death cause as a result of carcass decomposition condition, observer training and available necropsy facility. The careful definition of a standard baseline examination protocol to be agreed between stranding schemes is necessary to ensure large spatial coverage, large sample size and robust conclusions. To have confidence in both these point prevalences and any trends observed over time requires examination of a sufficient number of cases based on *a priori* power analysis. The following worked example demonstrates how the sample size is dependent on both the required precision of the result and the estimated prevalence of the cause of death in question.

Using data collected by the UK CSIP between 1996–2010, harbour porpoise show a overall mean of 21.4% of cases where the cause of death was anthropogenic, with a annual mean ranging between 11.4% and 34.5% and a three year average mean ranging between 29.3% in 1996–1998 to 13.6% 2008–2010 (Table 4.3). Power analysis suggested that in order to detect, at a power of ≥80% and α=0.05, a 15% change to the starting prevalence of 30% for anthropogenic causes of death, required a minimum sample size of 134 animals. For common dolphins, currently bycatch equates to anthropogenic cause of death (Table 4.4). Identifying a similar change of 15% in current mean common dolphin bycatch of 49.6% requires at least 183 animals. Identifying change in rarer conditions, such as gas embolism, or over shorter time spans will require considerably larger sample sizes.

These power calculations depend on the assumption that there is no change in the selection criteria during the period in question and the two time periods are independent. Providing these criteria are met, it is feasible to use strandings data to detect changes in different causes of death with some degree of confidence. However, obtaining sufficient sample size to detect small changes or over short time scales will require collaboration and harmonisation between European stranding and observer bycatch programmes.

Table 4.3. Harbour porpoise data from the UK stranding scheme (PME = post mortem examination).

Year	Number of PME	Three year running total for PME	Number of Anthropogenic causes of death	Three year running total for Anthropogenic causes of death	Three year percentage
1996	84		25		
1997	87		30		
1998	76	247	19	74	29.96
1999	80		20		
2000	77		20		
2001	118	275	16	56	20.36
2002	120		30		
2003	117		17		
2004	166	403	47	94	23.33
2005	128		26		
2006	102		15		
2007	56	286	11	52	18.18
2008	71		8		
2009	52		6		
2010	75	198	13	27	13.64

Table 4.4. Common dolphin data from the UK stranding scheme (PME = post mortem examination).

year	Number of PME	Three year running total for PME	Number of Anthropogenic causes of death (all bycatch)	Three year running total for Anthropogenic causes of death	three year percentage
1996	31		14		
1997	17		7		
1998	21	69	5	26	37.68
1999	14		4		
2000	23		10		
2001	37	74	20	34	45.95
2002	44		31		
2003	41		26		
2004	46	131	33	90	68.70
2005	25		14		
2006	21		12		
2007	16	62	7	33	53.23
2008	43		4		
2009	15		10		
2010	9	67	3	17	25.37

4.6.1 Other potential indicators from strandings

Dolphin demographic parameters can be estimated directly from repeated observations of marked individuals (Silva *et al*., 2009), but such longitudinal studies are difficult for pelagic populations with broad distributions like harbour porpoise, common dolphin and pelagic bottlenose dolphins. Reproductive parameters can also be estimated through the examination of biological samples provided by strandings or bycaught individuals (e.g. reproductive parameters; Westgate and Read, 2007; Murphy *et al*., 2009). Strandings represent the most extensive source of demographic data for cetacean populations living in regions susceptible to be reflected in stranding. In this respect, the analysis and modelling of drift patterns potentially allows biological information obtained from stranding to be associated with an area of likely origin (Peltier, 2011; Peltier *et al*., 2012). Survival and reproductive parameters can be estimated on the basis of age and reproductive state determined from teeth sections and gonad examination from stranded cetaceans (Westgate and Read, 2007; Murphy *et al*., 2009; Mannocci *et al*., 2012). These parameters can then be used as inputs in demographic models to conduct population projections and risk analyses as well as potentially providing indicators in their own right.

Survival can readily be estimated from dolphin age-at-death distributions, for instance by using standard life table calculations, Kaplan-Meier estimates or the Siler model (Slooten and Lad, 1991; Stolen and Barlow, 2003). In common dolphin, and possibly in most small cetacean stranding data in the ICES range, the age structure derived from strandings is likely to result from both natural and man-induced mortalities. Both approaches can be used to derive age-specific and average mortality rates of the sampled population. In addition, this mortality can be apportioned to different causes of death based on diagnosed causes of deaths from strandings. Absolute values must be treated with caution due to possible biases in the strandings data and due to the effect of varying age structure (the mortality estimates assume a stable age structure). However, trends are likely to be informative. Thus, time-series of such data could provide an indicator of changing population status, with increasing mortality potentially indicating an undesirable trend. For example, between 1992 and 2005, there was an apparent reduction in maximum age of stranded harbour porpoises in Scotland (Figure 4.14); this change in age distribution was statistically significant.

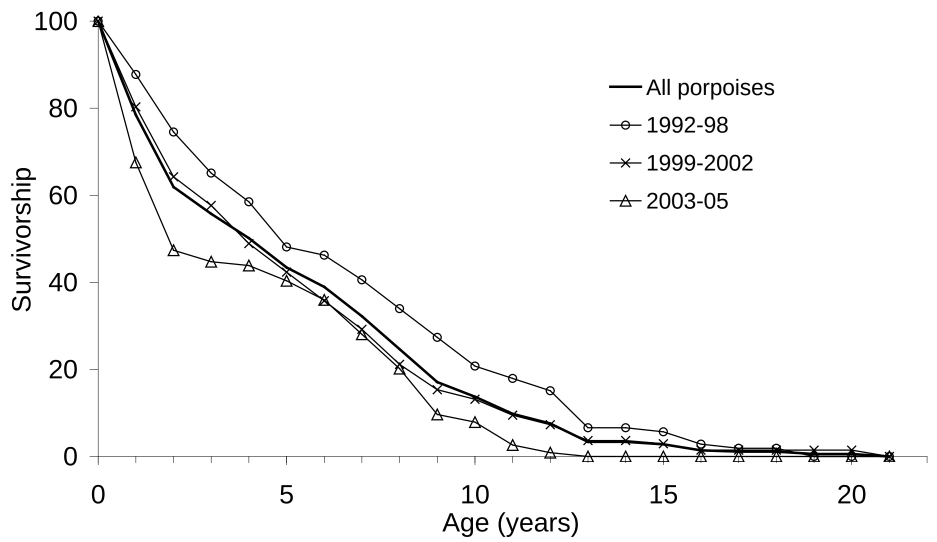


Figure4.14. Survivorship curves for harbour porpoises in three time periods between 1992–2005.

Using 1000 bootstrap runs, per sample size for the dataset suggests that an increase in mortality of >25% will be detected (Figure 4.15). Although this is rather large, the Scottish data show a 50% increase over the period 1999–2001, suggesting that it is sufficiently sensitive to serve as an MSFD indicator. However, an important caveat relates to interpretation. In a population in which age structure is shifting, observed mortality rate from strandings responds slowly to changes in the underlying overall mortality rate. In fact, as confirmed by results from simulations, the observed strandings rate is sensitive to changes in juvenile mortality but not to changes in adult or overall mortality (Figure 4.16/Pierce *et al.*, unpublished data).

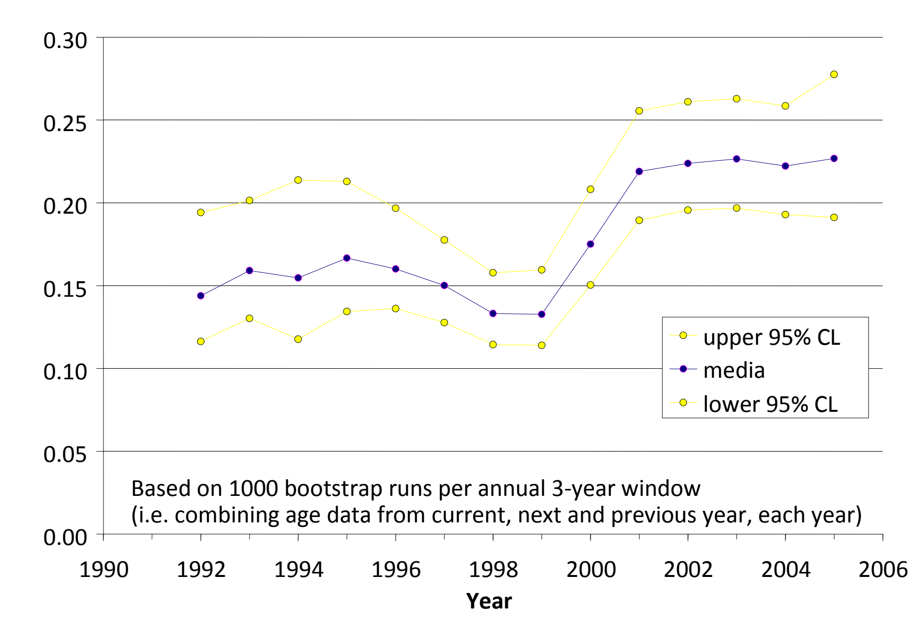


Figure 4.15. Change in harbour porpoise mortality.

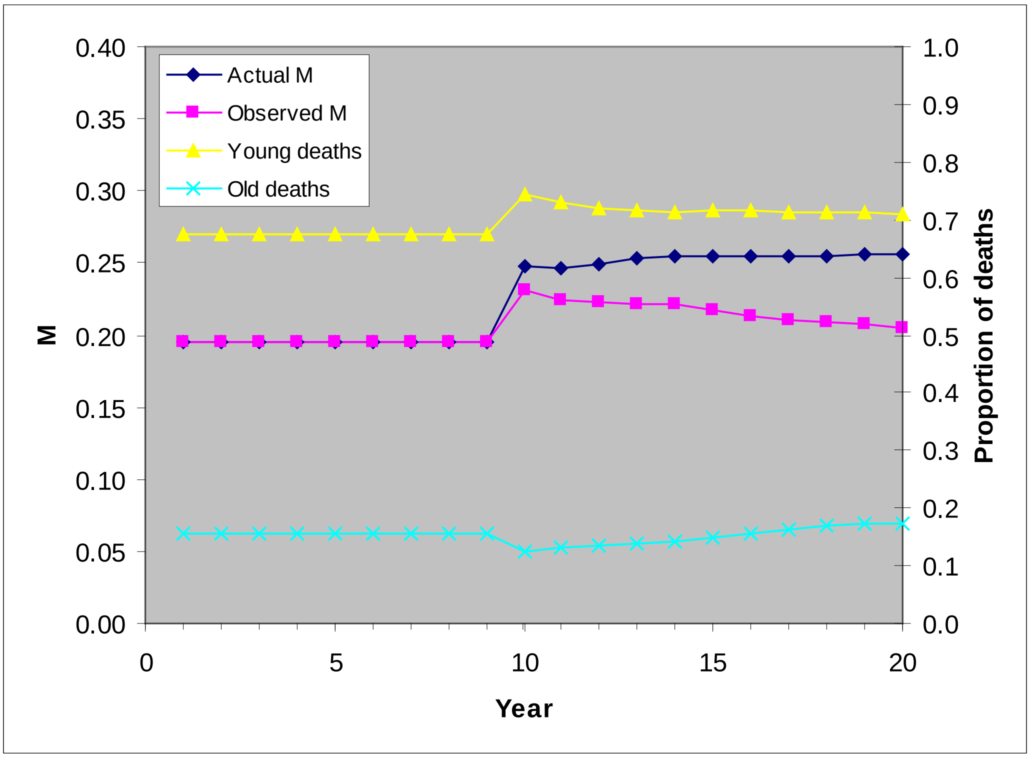


Figure 4.16. Observed mortality and underlying mortality in a simulated population subject to a doubling of juvenile mortality. Also shown are expected changes in relative numbers of young and old animals in strandings.

4.6.2 Synthetic compound contamination: PCBs

Marine mammals are exposed to a range of potentially toxic chemicals in their environment. Some lipophilic and persistent organic compounds bioaccumulate to very high levels, particularly in marine top predators such as harbour porpoises, bottlenose dolphins and killer whales. Detailed research on UK-stranded harbour porpoises conducted under the UK Cetacean Strandings Investigation Programme (www.ukstrandings.org) has shown strong links between elevated blubber PCB levels and infectious disease mortality (Jepson *et al.*, 1999; Jepson *et al.*, 2005; Hall *et al.*, 2006) consistent with fatal PCB-induced immunosuppression. In UK harbour porpoises, an initial decline in blubber Σ25CBs concentrations was observed in the mid-1990s, but has subsequently plateaued from 1997 to the present indicating that toxic impacts of PCBs will likely continue for some time (Law *et al.*, 2010).

In one case-control study of UK-stranded harbour porpoise the risk of infectious disease mortality increases 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. In a second case-control study of UK-stranded harbour porpoises, mean summed 25CB congeners in the "healthy" control group (death due to physical trauma) was 13.6 mg/kg lipid and 27.6 mk/kg lipid for the that died of infectious diseases (Jepson *et al.*, 2005). These studies allow an estimated threshold of toxicity (including immune suppression and reproductive impairment) for blubber PCB concentrations that exceed 20 mg/kg lipid wt (for summed 25CB congeners) in harbour porpoises. This equates to a blubber PCB toxicity threshold concentration of 13 mg/kg lipid wt (for summed ICES7 CB congeners) based on standard regressions between summed 25CBs and summed ICES7 CB congeners. This 13 mg/kg concentration for summed ICES7 congeners could be used for other marine mammal species (not just harbour porpoises) to assess populations that may at risk toxic effects at individual and population levels.

4.6.3 Applicability of indicators from stranding schemes

The sample requirements for strandings based indicators are generally quite large. Although it is potentially possible to scale current national monitoring and surveillance programmes to meet these needs, it is unlikely at this time. An international approach, combining the output from the various national schemes is required should any indicators be developed from stranding material. Such an approach will require harmonisation of protocols and a system for rapid exchange of information on new strandings, such as the international database proposed by ASCOBANS (Deaville *et al*., 2012) and recommended by WGMME (2010). This would allow for an ecosystem-wide surveillance system, providing a more rapid alert to transboundary mortality events and to share data to increase power of statistical analysis on species with a low incidence of stranding. See Annex 3 for examples of potential indicators based on strandings data (using the ICG-COBAM summary sheet format).

4.6.4 Potential indicators from biopsy data taken from free ranging cetaceans

Biopsy sampling has become a valuable tool used to acquire samples from cetaceans, providing data on genetics, prey preferences, foraging ecology, contaminant loads, and physiological processes (Noren and Mocklin, 2012). These samples may also be more representative of the population than samples collected from dead or live stranded animals that may be ill or emaciated. Such an approach is being considered as part of the development of MSFD indicators for the Mediterrean Sea (Fossi *et al*., 2012).

It is well known that the various cetacean species occupy different habitats and exhibit different patterns of movement. As such they could potentially act as sentinels. For example the bottlenose dolphin often forms small resident groups, which could be an indicator of various anthropogenic pressures on the coastal environment (Walton *et al*., 2007). Similarly, the striped dolphin, the most abundant odontocete in the Mediterranean region, is distributed in deeper offshore waters and could be a sentinel of the pelagic environment (Panti *et al*., 2011). Finally, due to its broad ranging seasonal movements across the whole basin, the fin whale could represent an integrated indicator of the whole Mediterranean area (Fossi *et al*., 2010).

Significant advances have been made in the development and application of nonlethal biomarkers from cetacean skin biopsies to diagnose “toxicological stress syndromes”. This is increasingly applicable given the known synergistic effects on cetacean health from multiple and diverse stressors, such as bioaccumulation of anthropogenic contaminants, infectious diseases, shipping, collision trauma, noise, food depletion and climate change. A range of diagnostic markers are becoming available which can be employed on biopsy samples, which include markers for molecular and gene expression, genotoxicity, immunological competence, nutritional status and presence of contaminants, exposure to contaminants and feeding ecology (Fossi and Marsili, 2011).

As also suggested by ASCOBANS (2011), WGMME recommend that current biopsy protocols are reviewed to investigate if more use can be made of this emerging technology, in order to gain useful insight into the health of free ranging marine mammals and potentially contribute to the assessment of GES in the European North Atlantic.

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5 ToR c) Outline and review the effects of wave energy devices on marine mammals and provide recommendations on research needs, monitoring and mitigation schemes

5.1 Introduction

There is an increasing drive to expand the use of wave energy generation devices across the ICES area. Wave energy converters (WECs) are, however, at a relatively early stage of development when compared to other renewable energy technologies. This is reflected in the lack of knowledge of effects that these devices might have on the marine environment, and therefore a lack of information available for environmental consenting. In 2010 and 2011, the WGMME reviewed the effects of construction, operation and decommissioning of offshore wind turbines and tidal-stream energy converters, respectively, on marine mammals and provided recommendations on management and monitoring (WGMME 2010, 2011). This report provides the equivalent assessment of wave energy converters and their effects on marine mammals.

There are numerous parallels between the various offshore renewable industries in terms of technological approaches to installation, maintenance and decommissioning of these structures. At the same time, the effects of these other industries on various aspects of marine mammal ecology have been investigated and can now serve to inform the assessment of potential risks to marine mammals associated with wave energy converters.

At present most WEC designs are at the test stage of full-scale prototypes while a few (see below) are at the levels of full-scale operational devices; there are currently no full-scale commercial developments of multiple devices (arrays). Accordingly, our knowledge of the potential interactions of marine mammals with these devices is limited, based on the first investigations and inferences derived from comparisons with other industries such as the offshore wind sector, fisheries, and oil and gas developments. Considering these factors, this report will highlight current research needs and important issues of regulation and management to be addressed in the coming years.

5.2 Features of wave energy converters relevant to marine mammals

Wave energy converters, in the broadest possible sense, work by absorbing kinetic energy from the water column. There are a wide variety of wave energy converters in development, for use in a range of marine environments (including at least partially onshore, in shallow coastal waters as well as deeper waters further offshore). These vary both in their basic energy extraction concepts and in their specifics, including water depth requirements, water column position, extent of surface piercing, methods of seabed mooring/attachment, deployment techniques, extent and velocity of exposed moving parts, size and seabed footprints, noise emissions, lubricants used and maintenance/decommissioning requirements (Scottish Marine Renewables SEA 2007; WGMME 2011). Although some environmental interactions, such as removal of the wave energy itself, cable runs, maintenance boat access, anchoring and fisheries exclusions are likely to be generic, it is anticipated that, given the variability between device types, the majority of issues relevant to marine mammals will vary depending on the particulars of the individual devices. These can be aggregated into several broad design categories (based on the descriptions by ECN 2012; EMEC 2012):

5.2.1 Surface attenuators

These devices float on the surface, oriented perpendicularly to the wave front. They capture wave energy by changing their shape in response to passing waves e.g. through flexing of connected parts, and converting the energy thus obtained for storage and transmission using hydraulics. These devices are typically designed for use in offshore areas, where they are deployed by means of a system of mooring cables and anchors to ensure a fixed position. Power take-off is typically achieved through a separate electrical cable system. Examples of such a device include the Pelamis™ wave energy converter (Pelamis Wave Power Ltd., 2012; Figure 5.1A) and the Dexawave™ device (Dexawave A/S, 2011; Figure 5.1B).

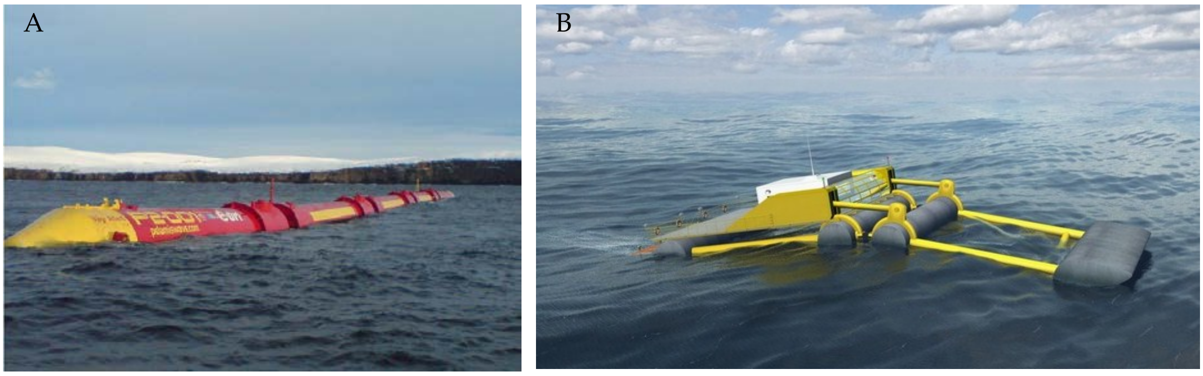

Figure 5.1. Examples of surface attenuators: A) © Pelamis Wave Power Inc. 2011. A Pelamis™ device at the EMEC test site, Orkney, UK; B) © Dexawave A/S 2012. An artists' impression of a Dexawave™ device.

5.2.2 Point absorbers

These devices, many of which resemble traditional buoys, also float at the sea surface but are designed to be able to obtain wave energy from any direction by means of vertical movement relative to a seafloor connector. Numerous different designs of this general type have been developed, including devices that convert kinetic energy to electricity at sea and then transport it to shore using subsea cables, as well as devices which pump water onto shore where the energy is taken off using conventional turbines. The connection between the surface float and the seafloor connector can be either flexible or rigid, depending on the device design. Examples of such devices include the PowerBuoy™ (OPT, 2011; Figure 5.2C), the WaveBob™ (WaveBob Ltd. 2012), and the WEC devices developed as part of the Lysekil project (Leijon *et al*., 2008). The Wave Star™ device consists of a series of floats, each connected by arms to a single rigidly moored platform (Wave Star A/S, 2011; Figure 5.2A). A very different approach, using a floating ring of interconnected air baffles, can be found in the AWS-III™ device (AWSOE, 2012; Figure 5.2B).

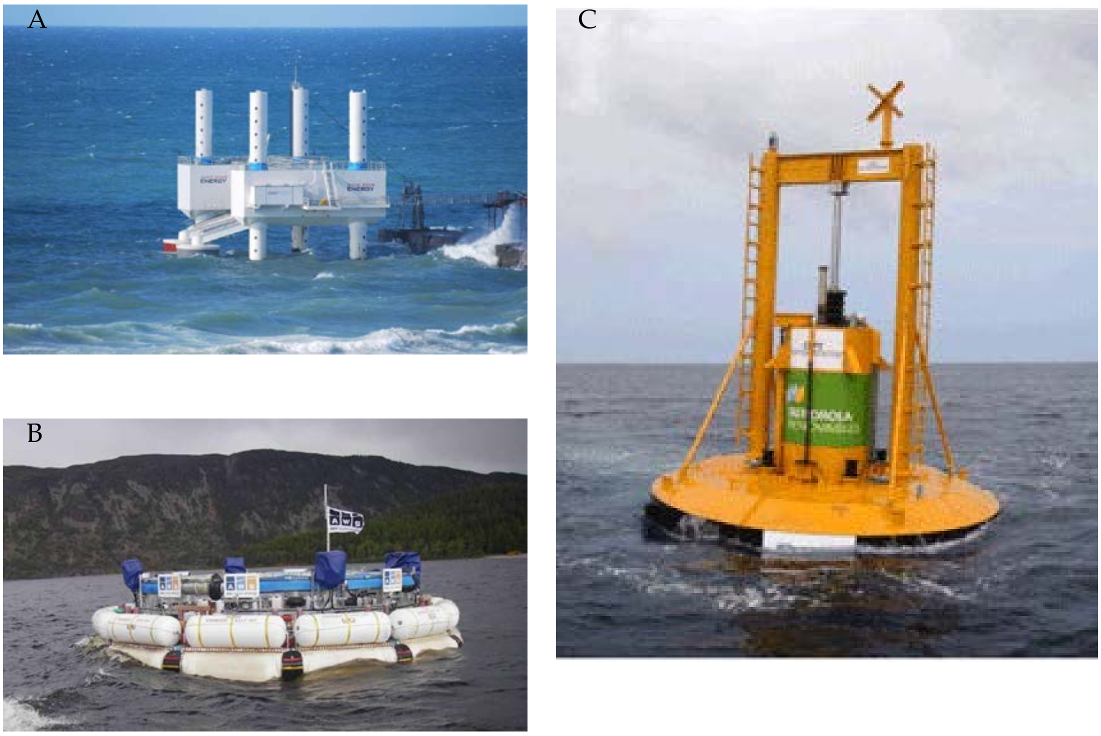

Figure 5.2. Examples of point absorbers: A) © Wave Star A/S 2011. The Wave Star™ 1:2 scale demonstration plant in Hanstholm, DK; B) © AWS Ocean Energy 2010. A 1:9 scale model of the AWS-III™ device being tested in Loch Ness, Scotland, UK; C) © OPT 2012. A PowerBuoy™ device.

5.2.3 Oscillating wave surge converters

These WECs take advantage of changes to movement of water affected by waves as they begin to surge and break while approaching shallower inshore waters. Generally speaking these devices consist of an arm which oscillates as a pendulum mounted on a pivoted joint in response to the movement of water in the waves. Energy can be taken off directly or transferred by pumping water onto shore to drive conventional generators. Their dependence on wave surge means that these devices can only be deployed in comparatively shallow, inshore waters. Examples of such a device include the Oyster™ (APL, 2012; Figure 5.3A) and the WaveRoller™ (AW Energy, 2012; Figure 5.3B).

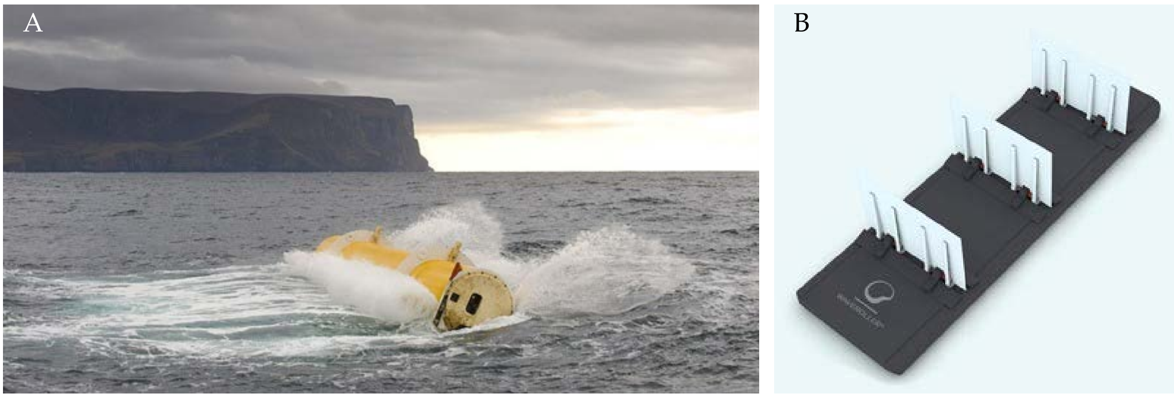

Figure 5.3. Examples of surface attenuators: A) © APL 2012. An Oyster device protruding from the water at the EMEC test site, Orkney, UK; B) © AW Energy 2012. A 3x 100KW WaveRoller prototype.

5.2.4 Oscillating water column devices

WECs of this type use wave energy to periodically compress air within a semi-enclosed space in response to wave-driven changes to water pressure. This generates air currents that drive turbines to take off the energy and convert it into electricity. Two broad types of such devices have been developed: devices built to operate within the intertidal zone (e.g. the Limpet™ which has been operating on the shore of Islay, Scotland, UK since 2000; VHW, 2012; Figure 5.4A), and floating devices that can be deployed further offshore e.g. the SperBoy™ (Embley Energy Ltd., 2010) and the OceanEnergy Buoy™ (OE, 2012; Figure 5.4B). Intertidally operating devices can also be incorporated into new or existing seawall or pier structures (e.g. the newly commissioned Mutriku wave plant in the Basque country, Spain; VHW, 2011).

Figure 5.4. Examples of oscillating water column devices: A) © VHW 2012. The Limpet™ Islay power plant, Scotland, UK. B) © OE 2012. The OceanEnergy Buoy™.

5.2.5 Overtopping devices

These WECs collect water from waves that spill over into a basin, from which the water can then be drained back to sea level (during which process energy can be taken off using turbines). Devices of this type may also contain elements which reflect waves towards the basin. One example of this type is the floating WaveDragon™ device (WaveDragon, 2011; Figure 5.5A) which can be deployed some distance offshore. Similar principles are used for land-based devices such as the Seawave Slot-Cone Generator™ (SSG; WAVEenergy, 2005; Figure 5.5B) that can be built into existing structures such as seawalls.

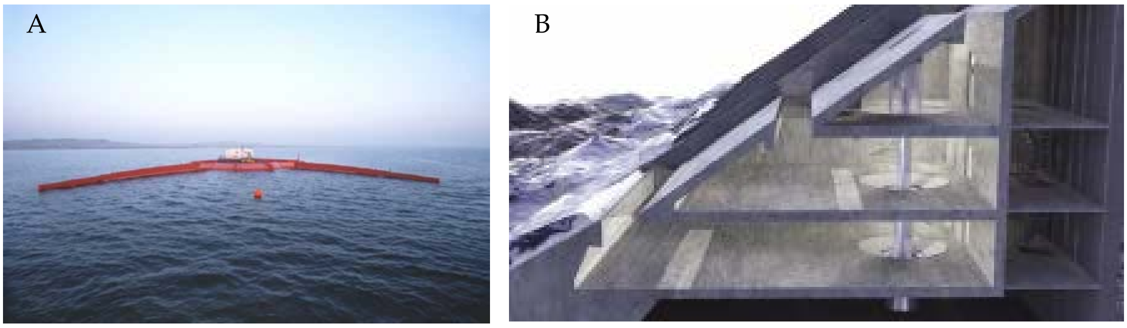

Figure 5.5. Examples of overtopping devices: A) © WaveDragon 2011. A WaveDragon™ device. B) © SSG 2005. Artists' impression of the Seawave Slot-Cone Generator™.

5.2.6 Submerged pressure differential devices

Devices of this type are deployed in comparatively shallow nearshore waters, where they take advantage of changes in water pressure as a result of successive wave troughs and crests passing overhead in order to generate power. This power can then drive water onshore through a system of pipes where the energy can be taken off using traditional turbines. The CETO™ (CWE, 2011a) device is an example of this type of system (Figure 5.6A).

5.2.7 Other

There are various other device designs that do not clearly fit one of the above broad categories. These include devices with flexible tubes that bulge in response to passing waves (the Anaconda™; Checkmate Sea Energy, 2012; Figure 5.6B) and devices using electroactive polymers to convert wave energy into electricity (SRI, 2008).

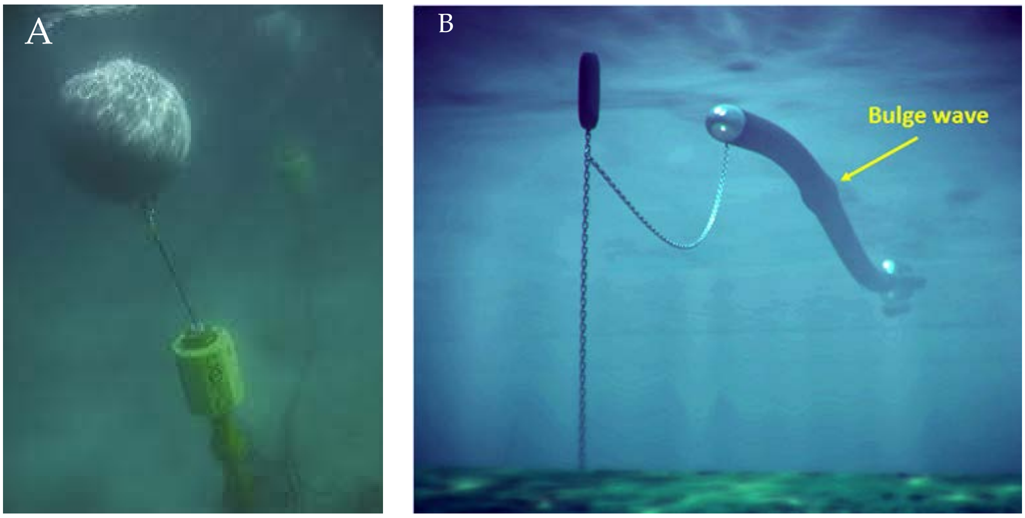

Figure 5.6. Examples of A) a submerged pressure device: © CWE 2011. A CETO™ device. B) an example of a completely different device design: © Checkmate Sea Energy 2012. Artist's impression of an Anaconda™ device.

5.3 General WEC characteristics

Most of the devices described here are still in an advanced prototype stage although some e.g. the Pelamis™, Oyster™ and Limpet™ devices are currently operational at a small scale. Individual devices have a generating capacity of between 10–750 kW; the ultimate goal is to create arrays of devices capable of generating energy at the 20–50 MW-scale. The operational lifetime of individual devices is generally expected to be in the order of 10–20 years.

Individual device designs vary greatly in terms of dimensions and inertial mass. For instance, most point absorbers are broadly similar in size and shape to large navigational buoys, whereas the Pelamis™ surface attenuator device has a length of 180 m in its current configuration.

Many devices (especially the surface attenuators and point absorbers) have some kind of surface expression as a critical operational element. These devices remain in place by means of a mooring system consisting of a range of elements such as anchors, cables and chain clumps, and are expected to be deployed some distance offshore. Other devices (such as the oscillating wave surge converters and several of the oscillating water column devices) need to be constructed in the surf zone or the intertidal zone for maximum results, and may require pile driving (the Wave Star™ device; Wave Star A/S 2011), pile drilling (the Oyster™; AMP 2012) or gravity-based systems (the CETO™ device; CWE 2012) to ensure firm attachment to the substrate. Exact installation and/or mooring methods are likely to vary widely based on specific device requirements, environmental characteristics and infrastructure availability. At the moment, most devices are designed for deployment in inshore coastal environments, although several (e.g. the Pelamis™) could potentially be deployed widely across continental shelf areas far offshore.

The moving parts of some devices (e.g. Pelamis™ and Oyster™) are directly exposed to the outside environment, but in many other devices the parts that take off the kinetic energy (such as a turbine) are either contained within the device (e.g.

WaveDragon™) or built on-shore separate from the device itself, with the energy obtained by the WEC being used to pump seawater through a more traditional hydro plant (e.g. CETO™). For those devices that do have moving parts exposed to the environment, movement rates of such parts are likely to vary according to differences in device design but are likely to be comparatively slow compared to other mobile marine devices (such as ship propellers) under normal circumstances. Elongated devices such as the Pelamis™ WEC or the Anaconda™ are expected to be somewhat responsive to changes in wave (and, to a lesser extent, wind) direction, while point absorbers are unlikely to be deflected. Other devices (e.g. Oyster™, Limpet™) are essentially immobile and attached to the underlying substrate.

There is currently limited knowledge of the sound output of the various WEC devices into the marine environment, although research in this field is rapidly advancing (e.g. SEA 2007). Sound from WECs can be generated by a number of different ways, including rotating machinery, flexing joints, structural noise (resonance of the device or its components), moving air, moving water, moorings, electrical noise, and noise caused by device instrumentation. Sound produced by these sources can enter the marine environment by direct contact with the water, mechanical transfer of vibrations within devices, or by transfer through the seabed or air (SEA 2007; Qinetiq Ltd. 2007).

Maintenance scheduling is likely to be device-dependent but will also depend critically on availability of suitable weather windows. The expected shortage of maintenance opportunities, and high costs involved in maintenance of any offshore marine structures, will provide a strong incentive for device developers to develop designs that require minimal maintenance. At least some devices will be periodically removed and returned to port for more comprehensive maintenance (e.g. AMP 2012; Pelamis Wave Power Ltd., 2012).

5.4 Summary of distribution and scale of some of the wave energy developments in ICES waters

This section provides a brief description of wave energy developments known to be either currently operating or in advanced planning stages within the ICES area, summarised by country. Device test sites and associated infrastructure are also included here.

Norway

The most important test site for wave energy in Norway is Runde Environmental Centre (REC), on the Norwegian west coast, where a 30 kW pilot oscillating wave column device has been operational since 2006 (REC, 2009).

Sweden

The main wave test site in Swedish waters is the Lysekil Wave Power Project on the country’s west coast, where up to 10 point absorber devices (each with a 10KW capacity) are to be installed by 2013 as an experimental array (Leijon *et al.*, 2008; Uppsala University, 2011).

Denmark

Since 2000, developers have been able to test devices at the Nissum Bredning test station (in sheltered Limfjord waters; NFRE 2011). In 2009 the DanWEC (Danish test site for Wave Energy Conversion) test center in Hanstholm (on the northwest coast) became available for device testing (DanWEC, 2011). To date, a 600 kW pilot version of the Wave Star™ point absorber device has been producing power at this site since 2010 (Wave Star A/S 2011), a Dexawave™ test plant was deployed in 2011 (Dexawave, 2011) and a 1.5 MW WaveDragon™ demonstrator device is being built for tests at this site (Wave Dragon, 2012).

United Kingdom

Wave energy development in the UK is rapidly advancing, with several devices currently operational and numerous other sites licensed for future development. This is further assisted by the presence of test centres at sea, particularly the European Marine Energy Centre (EMEC 2012) in Orkney (Scotland) and Wavehub (2012; Witt *et al*., 2012) off Cornwall (southwest England), where devices can undergo extensive sea trials under controlled, monitored conditions before being deployed commercially. On the southern Hebridean island of Islay (west coast of Scotland), a 250 kW oscillating wave column device (the Limpet™; VHW 2012) was activated in 2000 and has been supplying energy to the national grid ever since. It is currently mainly used as a test bed for further technical improvements.

In a world first, in 2010 the Crown Estate leased six areas of seabed under its jurisdiction (out to 12 nautical miles) in the Orkney/Pentland Firth area to wave energy developers (as well as five tidal energy sites; Crown Estate 2012; see Figure 5.7). The award of a lease guarantees the developer exclusive use (for energy production) of the site concerned, but consent from the industry regulator (Marine Scotland) still needs to be obtained for construction and use. To date, two developers are each testing a Pelamis™ device at the EMEC test site before being deployed in their respective leasing areas (E.ON 2011; SPR 2012). Pre-deployment feasibility studies of the other sites within the Orkney/Pentland Firth area (expected to involve at least Oyster™, AWS-III™ and Pelamis™ devices) are ongoing, with the six sites together expected to produce up to 600 MW annually if fully developed (Crown Estate 2012). Additional site lease licenses have since been awarded off Shetland (10 MW; Aegir Wave Power, 2012) and off Lewis in the Outer Hebrides (a combined total of 50 MW; Pelamis Wave Power, 2012; APL 2012). Since 2011, ocean trials of a single 150 kW PowerBuoy™ point absorber device have been ongoing in the Moray Firth (OPT, 2011). A further leasing round of selected sites off Northern Ireland came to a close in January 2012 (Crown Estate, 2012). Several applications for pilot projects off the coast of Wales are currently being developed (MEP, 2011).

Ireland

Scoping studies are currently underway to identify appropriate sites along the west coast of Ireland under the WestWave project (2012), which intends to install 5 MW worth of wave energy generating capacity by 2015. No final decision on device type has been taken at this point, but likely candidates include the Oyster™, Pelamis™, the OE Buoy™ and the WaveBob™ devices. The Galway Bay wave energy test site allows for device testing in a relatively sheltered location (MI 2012), with the development of additional test site capacity elsewhere along the Irish west coast currently being discussed.

France

The SEM-REV wave energy test site, located on the west coast of France near Nantes, became operational in 2010 and provides a test bed for wave energy technologies (SEM-REV 2012). No information is available on current or planned deployments.

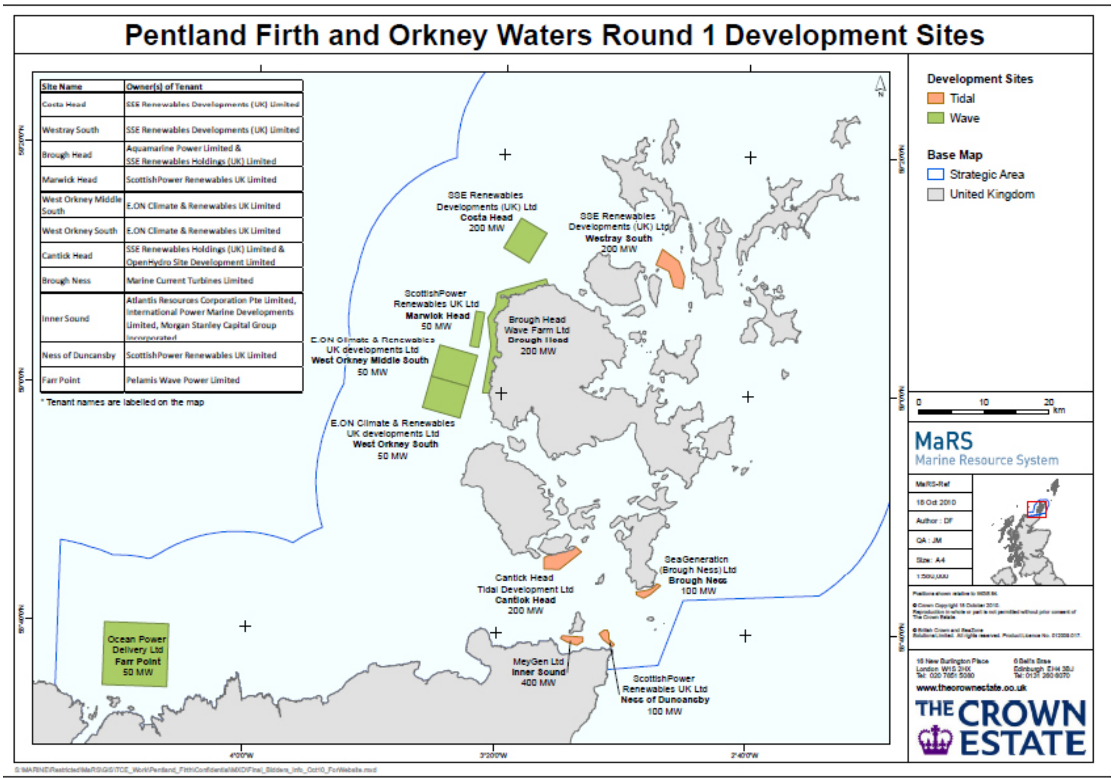

Figure 5.7. Crown Estate lease sites for wave and tidal energy development in Pentland Firth/Orkney waters, north Scotland, UK.

Spain

In mainland Spain, several wave energy projects are currently active, including the 300 kW oscillating water column system in the breakwater of Mutriku harbour (VHW 2011). Wave devices can be tested in the Basque country at the Biscay Marine Energy Platform (BIMEP) test site (EVE 2012) and in Cantabria at the Santoña test site, where a 40 kW PowerBuoy™ device has been deployed for further testing in anticipation of further development (OPT 2011). Several other sites along the Spanish Atlantic coastline are of interest for future development (Iglesias *et al.*, 2009; Iglesias and Carballo, 2010; APPA, 2011, Figure 5.8).

Off Gran Canaria, the Canary Islands Oceanic Platform (Plocan) is set to serve as a multi-industry testing facility by 2013 (Plocan, 2012); a 150 kW test point absorber has been tested here since 2011.

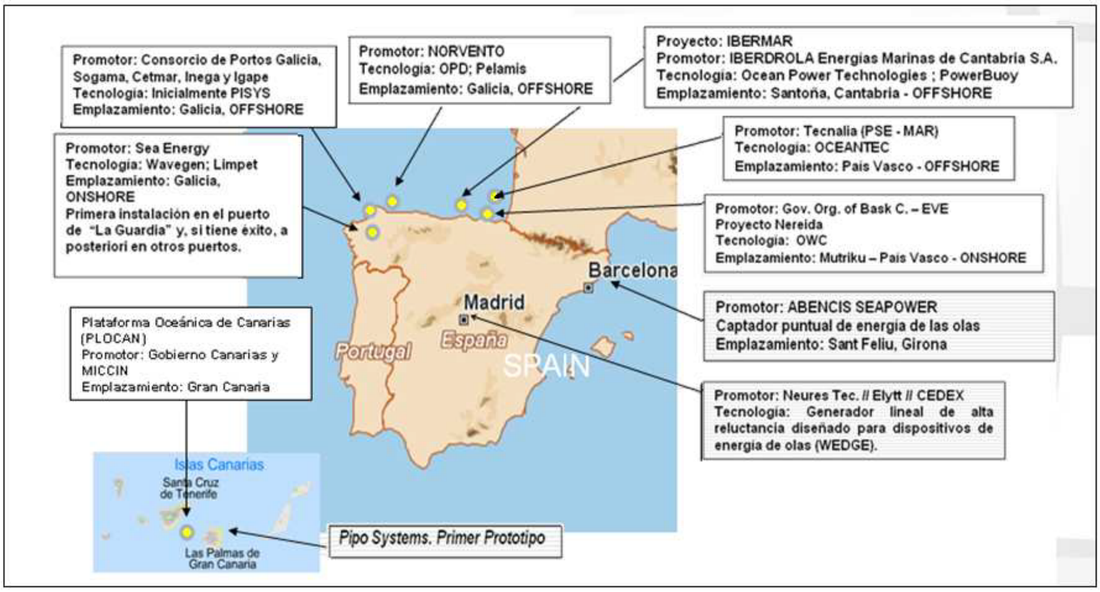

Figure 5.8. A summary of planned and ongoing wave energy projects in Spain (© APPA 2011).

Portugal

There are presently several marine energy testing sites in mainland Portugal (Dr. M. Ferreira, University of Minho, Portugal, pers.comm. 2012). Aguçadoura is a small testing site for renewable energy technologies (including wave and offshore wind) in the north of Portugal where several devices, including Pelamis™ devices, have been tested to date. San Pedro de Moel Zona Piloto is a larger testing site developed more recently in central Portugal, but no devices are in place at present. Apart from these two device testing sites, WaveRoller™ devices have been tested at the Peniche site (near the San Pedro de Moel site) since 2007 and a WaveRoller™ pilot plant is set to be installed there during the course of 2012 (AW Energy 2012).

In the Azores, a 400 kW oscillating water column device was installed in the harbour of Pico in 1999, which has since been used for testing and improving the technology (WEC 2011).

Canada

To date, Canadian interests in wave energy have tended to focus on the west coast of British Columbia, where wave fetches are larger than in the Atlantic. A demonstration facility is presently under development off the western coast of Vancouver Island, British Columbia, with a view to deploy up to 5 MW in CETO™ device capacity (CWE 2011b). In the northwest Atlantic, experimental devices have so far been tested on a small scale off Nova Scotia (WET 2007) and Newfoundland and Labrador (CNA 2011).

United States

Similar to Canada, most current wave energy interests in the US (including the development of testing facilities) have focused on the Pacific Ocean e.g. the Northwest National Marine Renewable Energy Center (NNMREC 2012) and the Hawai'i National Marine Renewable Energy Center (HINMREC 2011). To date, a single 40 kW PowerBuoy™ device has been operating since 2004 off Kaneohe, Hawai'i (OPT 2011), and an offshore wave test site is under development off Newport, Oregon (NNMREC 2012). Along the Atlantic seaboard, there have been tests of a 40 kW PowerBuoy™

device off Atlantic City (NJ), but to date no large-scale deployments of WECs have taken place (OPT 2011).

5.5 Potential effects of wave energy developments on marine mammals

Marine mammal species can potentially be impacted during installation, operation and decommissioning of wave energy devices in a number of ways. Similar to the 2011 WGMME review of tidal-stream energy devices, most of the effects described below are considered probable, but are speculative and require verification with any new device being built.

It is vitally important to realise that any effects suggested here are likely to be species-specific and will also be influenced by particular features of the development site. For this reason, extrapolation of any likely risk assessments from experiences with one species or area to a completely new species or area should only be undertaken after careful consideration.

5.5.1 Installation

5.5.1.1 Physical disturbance

There is a risk that marine mammals could be disturbed by the presence of installation vessels and equipment, particularly those that require deployment on or close to the shoreline. This is a particular risk for seals that are hauled out in these areas during their breeding period, as this could lead to temporary abandonment of young and potential subsequent increases in juvenile mortality. Moulting seals that are scared into the water may face increased energetic costs. In some cases where land-based infrastructure needs to be constructed in the vicinity of seal haul-out sites, there is a similar risk of disturbance as outlined above. Cetaceans in coastal and offshore waters may also be disturbed by installation activities as well as the continued presence of the WECs themselves. Neophobic individuals (or species) may be more likely to avoid devices at greater range, whereas other animals might actively choose to investigate devices more closely.

5.5.1.2 Underwater sound

As with other anthropogenic activities in the marine environment, wave energy extraction is likely to result in an injection of acoustic energy into the water. Throughout this document, the term “sound” is used to describe the acoustic emissions of devices, while the term “noise” is used to describe negative interference with the acoustic ecology of marine mammals. These negative impacts can include disturbance and habitat exclusion at considerable distances, as well as (at increasingly close range) masking of biologically relevant acoustic input from other sources, Temporary Threshold Shifts (TTS; temporary loss of hearing due to high sound levels) or even Permanent Threshold Shifts (PTS; permanent physical hearing damage as a result of high sound levels).

During site preparation, several predominantly acoustic methods are commonly employed. Depending on the method used (e.g. side scan sonar vs. seismic point surveys), several potential effects on the marine environment need to be considered. In the construction phase, some aspects of introduced underwater noise will have direct parallels with the offshore wind industry particularly when heavy lift vessels are used to deploy the devices and associated moorings and subsea cables (SEA 2007; Qinetiq Ltd. 2007; WGMME 2010). Acoustic disturbance of marine mammals due to installation of devices and cable-laying can occur both in water and in air (particularly for seals using haul-out sites). If the construction of infrastructure involves use of noise-intense procedures such as blasting, pile driving or drilling, the sound input without applied mitigation measures has a higher potential for impairment or even injurious impacts and wider displacement of animals (see WGMME 2010); this may also be relevant for onshore installations near the waterline. Only a very small number of WECs currently under development require the use of pile driving (Wave Star A/S 2011) or pile drilling (APL 2012), with the rest relying on anchors or gravity-based structures for stability. The acoustic impacts of pile driving on marine mammals are well understood with mitigation measures in place (Madsen *et al.*, 2006; WGMME 2010), but the impacts of pile drilling are much less well known (Nedwell and Howell, 2004). Several devices that use wave energy to drive seawater to a conventional onshore hydro-electric plant will require directional drilling through the substrate from shore in order to install the flow lines (e.g. the Oyster™; APL 2012), and sound emissions from this drilling activity also need to be considered.

Ships used for construction contribute to the ambient sound level in the area. Those producing lower frequencies pose the risk of masking of biological significant signals to marine mammals, thus effectively shrinking the range of their acoustic perception (WGMME 2011). This effect may be significant if ships used during deployment emit higher sound levels than ships normally occurring in the area and/ or if deployment occurs in areas that had heretofore seen very low levels of shipping activity (SEA 2007). Long-term effects of chronic exposure of marine mammals to anthropogenic noise are of considerable current interest but research in this field still remains limited (Tyack, 2008).

5.5.1.3 Collision risks

Vessels are needed for installation of tidal devices, their moorings and electric cables. As these activities are likely to take place in a stationary or slowly travelling mode, collision risk involving vessels during construction periods is likely to be lower than during commercial shipping activities (WGMME, 2011). Vessels involved in WEC installation need to be able to manoeuvre accurately at small spatial scales, which is typically achieved by using ducted propellers such as Kort nozzles or some types of Azimuth thrusters. Recently an increasing number of unusual seal mortalities have been reported in UK waters, consistent with injuries expected from animals being drawn through a ducted propeller (Thompson *et al.*, 2010), and concerns have been raised that harbour porpoises might be similarly affected (UKCSIP 2010; Dr. A. Brownlow, Scottish Agricultural College, UK, pers. comm., 2012). Such systems are used in a wide range of ships including vessels likely to be associated with wave energy developments, for example tugs, self-propelled barges and rigs, various types of offshore support vessels and research boats (WGMME, 2011). Although ships equipped with these propellers are not new, there has been an increase in the amount of operational time such ships spend in shallow inshore waters, partially driven by expansion of the marine renewable energy sector.

5.5.1.4 Reduced visibility

Increased turbidity leading to reduced visibility can occur during seabed installation of devices, cables and/or mooring components, with fine particles travelling even further from the disturbed area. It is conceivable that sudden, unexpected increases in turbidity may impact marine mammal foraging, social and predator/prey interactions. However, many marine mammals spend considerable amounts of time in turbid waters. Furthermore, WECs are likely to be deployed in comparatively energetic locations where large amounts of fine sediments are unlikely to accumulate, and any increases in turbidity as a result of resuspended sediments are unlikely to persist for any length of time (although this will need to be assessed for specific developments on a case-by-case basis).

5.5.1.5 Impacts due to contaminated sediments

Possible release of contaminants when dispersing sediments during cable and device installation could theoretically cause problems for marine species that are sensitive to contamination, such as marine mammals (WGMME, 2011). As described previously, it is likely that wave action will ensure rapid displacement of any contaminated sediment that might be resuspended, minimising risks to marine mammals.

5.5.1.6 Grid connections

Many of the most suitable sites for wave energy generation are in comparatively remote locations without suitable cable infrastructure connecting the devices to the national power grids (e.g. SEA 2007). This requires a potentially substantial investment in terms of additional interconnector cables, substations etc. Some of this infrastructure will be land-based but other elements will have to be deployed under water. This will require the presence of additional installing vessels and construction machinery outwith areas where devices are to actually be installed, and thereby increase the size of the footprint of the industry.

5.5.2 Operation

5.5.2.1 Collision

Marine mammals can be at risk of collision with the various different categories of WECs in various ways. For those devices meant for deeper water that have a surface expression e.g. surface attenuators and point absorbers, animals may potentially collide with the device itself while breathing, feeding, resting or travelling near the surface (Wilson *et al*., 2007). Collision risk is considered to be greater when a greater proportion of the device is below the surface (Boehlert *et al*., 2008). Devices may be less detectable under conditions of poor visibility (turbid waters), or reduced manoeuvring options such as in surge conditions or during storms. Animals could also potentially collide in mid-water with those devices that do not have a surface expression (e.g. the submerged pressure differential devices) as well as with interconnector cables or elements of the mooring system. It is worth noting that the mooring system of some devices can be quite extensive, relative to the size of the device itself (e.g. Pelamis Wave Power Ltd., 2012).

Marine mammals have the capacity to avoid and evade WECs, but only if they are able to detect the objects, perceive them as a threat and then take appropriate action at long (avoid, i.e. swim around) or short range (evade, i.e. dodge or swerve; Wilson *et al*., 2007). The ability of animals to detect devices depends on species-specific sensory capabilities, local visibility and level of sound output by the device relative to ambient noise levels. Neophobic individuals (or species) may be more likely to avoid devices at greater range, whereas other animals might actively choose to investigate devices more closely. There is presently no information on avoidance or evasive behaviour of marine mammals relative to wave energy devices, given the small scale of deployments to date. Detection distances are likely to be strongly influenced by ambient environmental conditions. Considering that WECs are moored to the seabed in sites that do not experience extreme tidal currents, it is likely that under normal circumstances animals should be able to detect the devices in time to avoid them, but that this may be affected by particular environmental conditions.

Finally, marine mammals are at risk of collision with vessels involved in device maintenance in the same way as described above under the Installation phase.

5.5.2.2 Sound emissions

The characteristics of sound emitted by WECs are likely to vary considerably between devices and also depend on the surrounding acoustic environment. To date, theoretical sound output of one surface attenuator-type WEC (the Pelamis™ device) has been independently reviewed (Qinetiq Ltd. 2007; SEA 2007). Since no direct measurements of Pelamis™ sound output were available, the review considered radiated sound data from similar machinery on board oceangoing ships. This comparison suggested that the machinery within a single Pelamis™ device (particularly the hydraulics) could generate sounds of 350 Hz at an intensity of up to ~140 dB re 1 μPa at 1 m. The review made a number of assumptions but suggests that "based on the limited data available, it is not expected that a wave energy device of this type (Pelamis™) would present any potential for causing PTS", and that "the risk of an animal experiencing TTS from a single 1 MW device of this type is insignificant" (Qinetiq Ltd., 2007). Results of array simulations furthermore suggest that "there is unlikely to be a significant PTS impact for commercial arrays of wave devices like Pelamis" (Qinetiq Ltd., 2007). Risks of temporary threshold shifts (TTS) appear similarly unlikely given the expected sound outputs. Behavioural reactions and masking are likely to occur over a limited range around WECs, but it is important that detailed impact assessments be carried out on a case-by-case basis for each individual project. Further in-situ work on assessing sound outputs from different devices under a range of environmental conditions is a necessary next step in assessing risks of widespread WEC deployment. During the operational phase, further sound is likely to be generated by vessels if devices are to be inspected at sea, or towed to port for servicing or repair.

5.5.2.3 Electromagnetic fields (EMF)

When in operation, electricity cables produce electric and magnetic fields, and concerns have been raised that these might affect the ability of seals and cetaceans to detect prey and/or navigate. There is, at present, no evidence that seals are sensitive to electromagnetic fields, although some large whale species appear to use variations in the geomagnetic field to navigate (Walker *et al.*, 1992) and passive electroreceptors have recently been described for one odontocete species (Czech-Damal *et al.*, 2011).

5.5.2.4 Contaminants

Parts of some WECs may need the application of antifouling products to retain functionality, although it has been suggested that bio-fouling is not likely to be a major issue for WECs (e.g., Langhamer *et al.*, 2009; APL, 2012; Pelamis Wave Power, 2012). Methods of achieving this have not yet been stipulated for many devices although antifouling paints are likely to be used. Further potential sources of contaminants include leaching of toxic compounds from sacrificial anodes or leakage of hydraulic fluids e.g., due to storm damage, corrosion, device malfunction or collision with vessels such as transiting (or accident-stricken, drifting) ships. The latter could lead to significant leaks of cargoes or fuel carried by the vessel involved (WGMME 2011). Because many WECs are intended to be moored in the open sea, they may be sought out by cetaceans for use as rubbing posts in a manner similar to ships or other structures, in which case direct skin-anti-fouling contact might possibly occur (Ritter, 2009; Williams *et al.*, 2009). Further details on the types of chemicals present on the outer surfaces of WECs and their associated infrastructure would improve the ability to assess the relative contaminant risks posed by these devices.

5.5.2.5 Habitat exclusion

It is unknown in what manner animals will respond to operating devices but, as with other anthropogenic activities, responses are likely to be species-specific. It is most likely that any behavioural reaction will be mediated by sounds emitted from the devices. While some animals may be attracted, neophobic species will likely show avoidance reactions to the novel structures and sounds. Such avoidance is unlikely to have a significant ecological impact unless it results in displacement and even long-term habitat exclusion (Wilson *et al.*, 2007; WGMME 2011). Any behavioural effects will be context-specific, i.e. could depend on factors such as age or reproductive state, behaviour and previous exposure (SEA 2007). Large arrays of WECs could potentially result in the loss of significant areas of habitat if animals do not perceive the gaps between the devices as passable based on the visual or acoustic signature of the array. Based on discussions with developers, typical array sizes are likely to be on the order of several km^2 for wave devices (7–100 devices; SEA 2007).

5.5.2.6 Downstream wave energy reduction

When wave fronts interact with WECs, there is likely to be at least a limited reduction in wave height downstream as a result of kinetic energy uptake by the WEC. To a certain extent diffraction of wave energy around the WEC will compensate for energy loss at the WEC, but some degree of downstream wave height reduction is still likely. Artificially reducing wave energy in nearshore waters may therefore impact geomorphological processes vital for maintenance of coastal environments, such as rates of erosion, sediment transport and deposition (Millar *et al.*, 2007).

Marine mammals could potentially be indirectly affected by these changes in a number of ways. Some animals might seek out calmer waters leeward of a WEC array for shelter, e.g., during storms, as has been suggested for harbour porpoises among aquaculture sites in Atlantic Canada (Haarr *et al.*, 2009). Calmer waters may mean that formerly-exposed rocky shores become more attractive as additional haulout sites for seals. Conversely, as wave action is one of the main drivers for longshore currents that carry sediments from which new beaches are rebuilt (Dean and Dalrymple, 2002), widespread extraction of wave energy might result in a decline in replenishment of beaches and sandbars downstream of device arrays, potentially threatening existing seal haul-out sites. However, it is likely that the largest waves will continue to bypass WECs without being significantly reduced, suggesting that their impact as an ecological driver will remain largely unchanged (Pelc and Fujita, 2002).

5.5.2.7 Physical restraint

Following a collision with power cables or mooring elements, marine mammals may be subsequently at risk of entanglement (Boehlert *et al.*, 2007). The entanglement risk posed by cables is dependent on their thickness (with thin cables providing a greater risk), their tension (with slack cables being more dangerous than taut ones), position in the water column (horizontal cables being considered more dangerous than vertical ones) and the materials chosen for their outer casing (smooth cables being less likely to entangle than rough ones). Entanglement risk involving cables is most likely to be a problem for larger cetaceans, particularly foraging baleen whales, but is not considered to be a major risk.

As a secondary effect entanglement may also be caused by lost fishing nets ("ghost nets") that may have become attached to sections of the WECs, and may thus impact small cetaceans and pinnipeds as well. There is presently no information available on at-sea ghost net abundance and distribution which would allow an assessment of the severity of this risk. WECs are not envisaged to have any effect on ghost net numbers, but may aggregate them if the nets become entangled by devices, cables or other infrastructure. If WEC array sites indeed act as Fish Aggregating Devices or otherwise lead to increased abundance of commercially targeted species (see Ecological Effects section below), it is conceivable that fishing activities seeking to exploit these species might become concentrated near these sites with increased entanglement risk to marine mammals. Alternatively, a shift of fishing effort out of an area due to WEC deployment may lead to changes in marine mammal bycatch in a wider area, in terms of absolute numbers and/or distribution.

There is a risk that seals or small cetaceans might enter the chamber of shore-based oscillating water column devices, and be unable to find their way out again, although this has not so far been observed in the Limpet™ device operating on Islay since 2000 (VHW 2012; D. Moysey, Marine Civil Engineer, Voith Hydro, pers. comm., 2012).

5.5.2.8 Injury through moving parts

Some of the WECs operate by means of moving parts that are exposed to the environment, such as articulation of segments of surface attenuators (Pelamis Wave Power, 2012), the flaps of oscillating wave surge converters (e.g., the Oyster™; APL, 2012) and even the turbines involved in power take-off in overtopping devices such as WaveDragon™ (WaveDragon ApS 2012; the latter being mainly a concern for seals that might enter the overtopping basin). If animals are unable to detect these moving parts in time to avoid them, there is the potential for injury by being struck by, or crushed between, these parts. These risks would presumably be exacerbated under conditions of poor detectability and/or when device movements are likely to be faster than average, e.g., during storms (Wilson *et al*., 2007).

5.5.2.9 Ecological effects

Widespread deployment of WECs in inshore and offshore waters, as currently proposed for some areas, has the potential to impact marine mammals in a range of indirect ways, by changing the local environment. Devices with a surface expression (e.g., the Pelamis™) could become attractive as haul-out sites for seals (Boehlert *et al*., 2008; Nelson *et al*., 2008). This might allow for a local expansion of foraging ranges for individual animals further offshore, although it might also put animals at greater risk of collision or injury if devices contain moving parts exposed to the outside environment.

All WECs, particularly those intended for deployment in deeper, offshore waters, are likely to alter their immediate environment. Many of the mooring systems currently under consideration are designed to operate on sediment rather than exposed bedrock, and offshore WECs are most likely to be deployed over areas of sediment. The introduction of hard substrate into this type of environment (the WEC device itself, but also associated mooring and cable elements) will lead to the appearance of communities associated with hard substrate, while the sedimentary communities within the immediate mooring footprint may be damaged or destroyed (Langhamer and Wilhelmsson, 2009; Langhamer *et al.*, 2009; but see Langhamer, 2010 for a consideration of natural variability). To date the evidence suggests that some species associated with hard substrates might become more abundant in the immediate vicinity of WECs and their moorings, both through colonising the devices and moorings themselves and through generating increasing amounts of hard shelly debris in the sediment surrounding the WEC, facilitating further settlement of hard-substrate species. These processes could result in locally elevated levels of prey biomass (particular benthic fish species) that may attract marine mammals, in a manner similar to other hard structures (Todd *et al.*, 1999). Furthermore, many different fish species are attracted to floating objects, a phenomenon that has long been exploited by fishermen worldwide through the use of Fish Attracting Devices (FADs) (Fonteneau *et al.*, 2000; Castro *et al.*, 2002). WECs floating at the surface may thereby inadvertently act as FADs leading to an increase in fish abundance, potentially resulting in locally elevated levels of prey biomass that may attract marine mammals (e.g., Brehmer *et al.*, 2011) as well as other piscivorous species, although this may subsequently also attract top predators such as sharks or killer whales. The closure to fishing of areas immediately surrounding WECs may also contribute to changes to local productivity and biodiversity, with possible knock-on effects for marine mammals (Inger *et al.*, 2009). Attraction of marine mammals to devices may also put them at greater risk of collision or entanglement in cables. Nonetheless, both individual WECs and WEC arrays may provide suitable foraging habitat for some marine mammal species as well as provide some form of refuge from vessel traffic and specific types of fisheries.

5.5.3 Decommissioning

Current device deployment plans suggest a device operational lifetime of 10–20 years, after which the device operator is likely to be required to remove the device and all associated infrastructure according to specified decommissioning standards (UNCLOS, 1982; DECC, 2011; SEA, 2007). In the UK guidelines, it is recognised that under certain circumstances (including when "the installation or structure will serve a new use, such as enhancement of a living resource, or serves a purpose beyond that of renewable energy generation, and would not be detrimental to other aims such as conservation") complete removal of devices may not be the best solution (DECC, 2011; SEA, 2007). The importance of WEC-related infrastructure for marine mammals needs to be periodically reviewed to ensure that eventual removal of this infrastructure will not have detrimental effects on particular marine mammal populations.

Decommissioning of WECs is likely to involve structure/device removal, waste and debris clearance and disposal, seabed restoration and subsequent maintenance, monitoring and management of the site (SEA, 2007). Many of the activities involved with these steps are similar to those encountered in device installation, and as a result many of the associated risks to marine mammals (e.g., collision with maintenance vessels, noise, seabed disturbance, and disturbance of animals) are also broadly similar. Removal of elements of the mooring system and other submerged hardware may pose the greatest impact risk, particularly if structures such as piles need to be physically removed from the seabed. In UK waters, pile removal is likely to involve excavating the entire pile or cutting off the exposed parts, with the use of explosives unlikely to be approved except under exceptional circumstances (SEA, 2007; DECC, 2011).

5.5.4 Synergistic effects

Most marine mammals are highly mobile and are therefore likely to spend only a small proportion of their time within the effect range of a single wave energy converter or even within an array of these devices. The effects of a WEC array could potentially be more severe if it were sited in specific areas of habitat of vital important to particular populations or species of marine mammals. When passing through multiple areas with WECs or other marine infrastructure animals will also be exposed to a variety of stressors, varying widely in their nature and impact. Those stressors can impact the animals directly (i.e. first order effects) or impact animals as secondary order effects (e.g. by changes in abundance of prey). The numerous potential ways in which such multiple stressors can interact remain poorly understood.

The deployment of wave energy devices is but one of many concurrent activities that might take place within a given marine area. As this sector is likely to expand (both geographically and in terms of numbers of devices) in the coming years, it is important to consider what kinds of interactions might occur with other marine industries and how such interactions might affect marine mammals. Several such potential impacts involving fisheries have already been noted. Placement of WEC arrays and subsea cable infrastructure in previously fished areas may result in improved conditions for those marine mammals that choose to enter such areas once they are closed to certain types of fishing, but could result in displacement of fishing effort leading to changes in the wider spatial distribution of bycatch. Appropriate marine spatial planning is essential in order to avoid or minimise such conflicts and their potential negative effects on marine mammals.

5.6 Overview of international guidelines on monitoring

In 2010, the WGMME undertook a review of general national and international guidelines and regulations for monitoring of marine mammals, and those focused on monitoring and mitigation of the effects of offshore wind farms, many of which may also be relevant to WECs (WGMME 2010). The wave energy sector is not as well developed as that of the offshore wind industry and, consequently, nor are the national guidelines.

The majority of the recommendations provided in WGMME (2010) on baseline and impact monitoring of offshore wind farms are relevant to wet renewables (including wave energy extraction). Understanding how marine vertebrates perceive, avoid and evade wave energy devices needs to be assessed. In addition, quantification of the potential rate of collisions and the population level consequences of individual physical injury, arising from collisions, and habitat exclusions are required (Wilson *et al.*, 2007).

Given the difficulties of inferring animal interactions with wave energy devices from other anthropogenic marine structures and the obvious problems of experimenting with WEC devices on captive animals, it is currently difficult to empirically test the risks of many of the potential marine mammal-device interactions. It is therefore essential that full advantage is taken of test deployments and early arrays to gather information on the actual interactions between devices and wildlife. To this end, the Scottish Government has developed a Demonstration Strategy in which research activities, in addition to monitoring required as consent conditions, are undertaken to validate and improve the knowledge base for later licensing decisions.

On behalf of Scottish Natural Heritage (SNH) and Marine Scotland (MS), SMRU Ltd. and Royal Haskoning recently produced draft guidance on survey and monitoring of marine mammals (as well as other species of interest) at marine renewables development sites (e.g. Macleod *et al*., 2011; Sparling *et al*., 2011). This draft guidance is likely to form the basis of regulatory monitoring requirements within Scottish waters in the future.

In order to satisfy national and international requirements (e.g. the Habitats Directive), monitoring schemes need to gather baseline information before construction begins, as well as continued impact monitoring during the construction, operation and decommissioning phases of the deployment. Broadly, monitoring must take place at spatial and temporal scales that are appropriate to assess impacts upon marine mammals at the population level. The following broad questions (based on Macleod *et al*., 2011) are suggested examples of issues that monitoring programmes need to address:

- Do marine mammals occur in the area of interest?
- What is the spatial and temporal distribution and abundance of marine mammals in the area?
- What are the marine mammals using the area for? (e.g. foraging, breeding)
- What is the sensitivity of marine mammals to different stressors linked to the construction, operation and decommissioning of WECs?
- Is detected change limited to the development footprint or over a wider area?
- Does the impact change with time or distance?
- Could any change at the population level be attributed to the development's construction, operation or decommissioning?
- Could any impact affect the conservation status of the population under (inter)national legislation?

5.7 Key knowledge gaps

With the increasing plans and efforts to deploying wave energy converters along the coastlines bordering the ICES areas in mind and with the lessons learned from the development of the offshore wind industry it is essential to develop consistent approaches for a thorough management of the potential effects this new technique might have on the marine environment in general and marine mammals in particular. Special emphasis needs to be put on protected areas and species suspected to be especially sensitive to pressures resulting from construction and operation of wave energy converters.

There is, however, still a lack of information and data describing the stressors produced by wave energy converters. This, along with the incomplete understanding of marine resources and coastal zone dynamics, introduces substantial uncertainty into the assessment. There are additional unanswered questions with regard to the cumulative effects of multiple WEC units arranged in arrays, and the ways in which these arrays might interact with other users of the marine environment. WGMME has assessed the effects of the construction and operation of offshore wind farms (2010) and tidal turbines (2011) on marine mammals, and many of the data gaps identified in these reports are also relevant to wave energy converters.

5.7.1 Baseline information

Many data gaps remain in our knowledge of basic biological features of many marine mammal species (e.g. spatiotemporal distribution, population size and structure, foraging and breeding areas) as well as any effects of WECs on these species. Filling these data gaps is likely to be the responsibility of academic institutions and national regulators, rather than individual developers. In the context of WECs, the following data collection needs are of highest priority:

- Abundance, seasonal distribution, migration patterns, population structure and development and habitat use for all marine mammal species in the areas of interest.
- Information on diet and foraging ecology for all marine mammal species in the areas of interest.
- Assessment of behavioural interactions of marine mammals with wave energy converters.

5.7.2 Noise

Underwater noise can be generated by a variety of sources in conjunction with the site preparation, construction, operation, and decommissioning of wave energy devices. Their potential effects on marine mammals can be diverse and an assessment can be complex (NRC 2005; Boehlert *et al.*, 2007; Southall *et al.*, 2007; Ellison *et al.*, 2012). Underwater sound plays a primary role for marine mammals. However, the acoustic sensitivity of many marine mammal species remains poorly studied or completely unknown. Therefore the analysis of impacts needs to be relevant for the species found near these devices and needs to be related to the specific sounds emitted by each particular type of WEC system. It has to be stressed that the range of sounds generated by these devices remains as yet largely undescribed.

The sounds emitted during the construction, operation and decommissioning of the systems will have to be assessed separately. Depending on the method used to install the devices intense noise can be generated and emitted into the marine environment, with particular construction methods such as blasting or pile driving of particular concern. During the operational phase of WECs onshore systems are likely to resemble natural sounds and emit only low levels of additional sound, while offshore installations may produce mechanical sounds at higher intensities which, depending on the frequency, may have a considerable impact on marine mammals.

Dedicated studies need to be conducted to document the acoustic characteristics of sound emitted during site preparation, construction, operation, and decommissioning of single devices. This should also take into account the sounds emitted by the ships used during installation and cable laying as well as the potential cumulative sound field of an array of WECs.

Currently the key data gaps in terms of sound emissions are:

- Acoustic measurements need to be conducted for the various techniques used during installation of the devices.
- The acoustic signature (level and spectrum as well as their temporal variation) of single WEC devices and multiple systems in an array needs to be monitored. These measurements need to be conducted both inside and outside of the array. This is especially important to address the generation of synchronous or asynchronous noise by an array.
- Ambient (background noise) needs to be monitored in a wide variety of environmental conditions, with particular focus on higher sea states.
- Measurements need to be gathered under different sea states conditions to differentiate background noise from the device noise, and to understand

how noise generation changes under different environmental conditions (e.g., by means of seafloor-mounted passive acoustic recorders).

- Sounds emitted by the ships employed during the installation/decommissioning process or for maintenance need to be measured under different environmental conditions.

Most of these data gaps could be filled by site developers as part of the regulatory licensing process. Ambient noise monitoring is also likely to be initiated as part of wider environmental monitoring efforts under the EC Marine Strategy Framework Directive (Tasker *et al.*, 2010).

There are also several data gaps concerning acoustic aspects of animal-device interactions:

- Auditory studies need to be undertaken to test the acoustic sensitivity and acoustic tolerance of those marine mammal species that are at risk but where data are currently unavailable to assess ranges of auditory perception. This is particularly important when construction of the WECs involves the emission of intense sound into the underwater environment.
- Auditory studies are needed to investigate the potential for masking of communication and/ or other biologically significant sounds by means of sounds emitted by WECs.
- Behavioural studies including controlled exposure studies on free-ranging animals could be conducted.

These particular data gaps are likely to be more appropriately addressed at a broader level by academic or regulatory bodies.

5.7.3 Electromagnetic (EM) fields

Large marine generators and the high voltage alternating and direct current cables that transmit power between devices and the land have the potential to interact with aquatic animals that are sensitive to electromagnetic fields. Although this is known to affect some fish species there is currently little understanding of whether or how EM fields might affect marine mammals. Further basic research of the interaction between marine mammals and EM fields (generated by marine renewable devices as well as other marine industries) is therefore necessary:

- Dedicated studies on sensitivity of marine mammals to electromagnetic fields and their behavioural reaction to anomalies in these fields are required.
- Baseline studies need to be undertaken to map naturally occurring EM fields at sites of interest before and after deployment of energy-generating and transmission equipment. This will allow an assessment of possible relevance of superimposed magnetic fields from the electrical currents produced and transmitted.
- The EM output of marine renewable devices needs to be assessed with and without electrical current production. This would allow an assessment of any impacts on marine mammals under different scenarios. It is important to address the difficulty of separating the effects of multiple impact agents (stressors). This approach would probably require the ability to switch from transmission to dummy loads.

5.7.4 Cumulative effects

WECs can impact on marine mammals in a number of different ways, and the interactions between these different types of impacts are at present only poorly understood. Further basic research is needed to clarify these interactions.

There are likely to be differences in the way marine mammals respond to individual devices, as opposed to when multiple devices are deployed in arrays over larger areas. In the absence of robust information on the impact of single devices it is of course even more difficult to quantify the impact of arrays. Further work is necessary to compare and contrast (first through modelling) the likely effects of different array configurations on marine mammals, in relation to a range of environmental parameters (e.g., bathymetry, current direction, distance from shore etc.). Attention needs be focused on studying the likely effects of different WEC array configurations on marine mammals, including sound output and potential barrier effects.

Cumulative effects may also occur due to interactions between WEC devices/arrays and other marine resource users, which also require further study. Much can likely be learnt from previous experience in the offshore wind industry.

5.7.5 Management

There remains considerable uncertainty surrounding the likely extent of negative interactions between marine mammals and wave energy devices, and the best method for incorporating such interactions into conservation management of marine mammal populations (WGMME 2011). Several management methods have been developed over the years (EC 1992; IWC 1999; IUCN 2001; HELCOM 2006; Hammill and Stenson, 2007; US NMFS 2007), but these methods differ in their basic assumptions and goals (Lonergan, 2011; 2012; Cooke *et al.*, 2012). Appropriate metrics do need to be developed to regulate any population-level deleterious effects on marine mammals of anthropogenic pressures in the marine environment, including marine renewable energy developments. To achieve this, target population size should be explicitly chosen and all appropriate data should be used to assess allowable impacts. The decisions on how high the target population levels should be set, and what level of impact (by whatever activity) is acceptable, needs to be made by society at large on the basis of the best possible advice provided by the scientific community.

5.7.6 Standardisation and data sharing

The use of standardised methods for conducting and reporting on monitoring at both small and large scales, as well as the provision of data collected through these efforts, are prerequisites for a sensible assessment of the effects of WECs on marine mammals. For commercial reasons the data collected by developers for most developments typically remain unavailable to the wider research community. In order to make better use of these datasets, a shared international common database could be set up that would allow wider dissemination of relevant datasets while ensuring confidential and anonymous treatment of commercially sensitive information. This would require standardisation of data, including metrics for reporting on individual test parameters but also study design, i.e. pre- and post-deployment monitoring as well as defining the requirements for reference sites. The Joint Cetacean Protocol (JCP), which was set up to aggregate and integrate information on cetacean distribution, abundance and population trends for the European North Atlantic using data from academic, government and commercial sources (to data, mainly offshore wind farm operators) working within their waters, provides an example of successful collaboration of the kind suggested here (Paxton *et al*., 2011; JNCC, 2012).

5.7.7 Animal-device interactions

As with tidal turbines (WGMME 2011) the diversity in technical design, size and developmental stage of wave energy converters is immense and the industry is evolving quickly. It is important to reiterate that the conduct of impact monitoring by site developers needs to be a condition upon any consent given for a demonstration device or array, taking into account what might be considered appropriate monitoring levels given environmental conditions and statistical data requirements in order to draw firm conclusions.

Animal-wave energy converter interactions are likely to be species-, site- and device- (or device-type-) specific, and therefore care needs to be taken when extrapolating conclusions about environmental impacts between species, sites and device types. Such extrapolation might eventually become justifiable once more insight into the stressor-response functions between all parameters is achieved. Many potential impacts described above are, however, likely to be relatively rare events. However, if scaled up to large numbers of devices, such rare events could still have a considerable impact. Under these circumstances, further in-depth investigation of such issues with a small number of device types could significantly advance our understanding of the risks posed by WECs more generally.

5.8 Recommendations

5.8.1 Main recommendation: Management framework

The marine renewable industry is developing rapidly and regulators need to make decisions on granting consent for licensing in the near future. As the industry expands from a few sites to a large number of sites over larger areas of sea, it will become increasingly important to be able to predict population effects in order to meet management objectives such as Favourable Conservation Status under the Habitats Directive and Good Environmental Status under the Marine Strategy Framework Directive. A good management framework requires a sufficient level of basic understanding of animal-device interactions including a deploy and monitor strategy for assessing these interactions; it would also benefit from ongoing data collection (monitoring) at appropriate scales to allow the incorporation of a feedback mechanism and to enable determination of whether management actions are allowing objectives to be met.

> The WGMME recommends the development of an appropriate precautionary management framework for marine renewable energy technologies.

This top-level recommendation includes several other, more focused recommendations:

5.8.2 Strategic spatial and temporal planning

In the absence of robust insight into the potential effects of WECs in the marine environment it is essential to take into account important areas to marine mammals (e.g. haul-out sites, breeding areas, designated sites of conservation importance) as well as sensitive periods (e.g. reproduction period). The assessment of possible effects and management of marine mammal populations needs to be conducted at biologically relevant scales. A strategic approach on a national and, where relevant, an international level should, where possible and relevant, encourage developments in less sensitive areas and periods while discouraging such activities in areas of greater importance for marine mammals (e.g. Natura 2000 sites).

> Whilst it is recognised that marine mammals are widespread, with an almost ubiquitous occurrence, WGMME recommends that a precautionary approach is taken to the placing of deployments, enabling the risks to be taken into account in the early stages through a deploy and monitor type approach.

5.8.3 Data gaps

Significant data gaps presently limit greater understanding of potential impacts of marine renewable energy devices on marine mammals. These include basic knowledge of marine mammals and how they behave around devices, emissions from different devices (e.g. noise, EM fields) and their effects on marine mammals under different environmental conditions, and cumulative effects of multiple devices in arrays. These data gaps need to be addressed to help reduce the impacts of marine renewable industries on marine mammals. Some of these gaps will need to be addressed by individual site developers, while others are best tackled by academic institutions or regulatory agencies. A collaborative approach between different stakeholders may be appropriate.

> The WGMME recommends that data gaps identified in this review be addressed. These include interactions with the devices, noise outputs under a range of environmental conditions, and synergistic effects of arrays versus individual devices.

5.8.4 Monitoring

Current monitoring efforts of distribution and habitat use of marine mammals, in relation to environmental impact assessments, e.g. for marine renewable energy developments, typically take place at far smaller spatial scales than are ecologically relevant for marine mammals, and are often undertaken independently without broader coordination. This results in numerous disparate datasets that are difficult to integrate when assessing overall impacts of marine renewable energy developments.

A nested monitoring approach, in which small scale monitoring efforts are developed in such a way as to allow integration with regularly repeated large-scale cross-boundary marine mammal surveys, would provide information at a spatial and temporal scale relevant to marine mammals while allowing the assessment of individual development sites. To enhance the power of the results all such monitoring efforts should be coordinated between adjacent developments and between countries sharing transboundary populations. Survey methodology should be standardized as much as possible, using surveying methods appropriate for the areas and species of interest, and results should be analysed as a whole (as exemplified by Ireland and the UK's JCP programme).

> WGMME recommends a cooperative monitoring approach for marine renewable energy developments is taken, which combines small scale monitoring efforts with large scale cross-boundary marine mammal surveys in order to provide information at a spatial and temporal scale relevant to marine mammals.

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6 ToR d) Update on development of database for seals, status of intersessional work

As there were no further developments on the ICES seal database during 2012, this Terms of Reference was deferred until next year's meeting.

WGMME (2010) reported that there was a requirement to collate information from seal population monitoring programmes across the ICES Area and to populate a database so details on harbour and grey seal populations in different regions/countries can be more easily compared. Despite the lack of development with this ToR over the last two years, the need for such a database is becoming more imperative with European legislation requiring the development of indicators (see ToR b).

WGMME (2010) noted that '*the longevity of this seal database is entirely dependent on the frequency and extent to which it is populated with information from different countries. Many organizations that monitor seal populations are, understandably, very protective of their data, as it takes a lot of time, expense and effort to collect and collate. It is imperative that the database remains secure and that its contents are not accessible by anyone without the consent of the contributors. Some data are available annually on the Internet (e.g. Wadden Sea Trilateral Seal Expert Group http://www.waddensea-secretariat.org/QSR-2009/20-Marine-Mammals-(10-03-05).pdf; UK Special Committee on Seals http://www.smru.st-and.ac.uk/pageset.aspx?psr=411 for annual reports).' This situation has not changed.......To date, Denmark, Germany, the Netherlands, Belgium, Ireland, Sweden, Norway and the UK have provided data. Data from France have been requested but may be problematic as they are collected independently by a number of different organizations and are not collated by one Governmental authority. Scientists in La Rochelle are attempting to collate the relevant information. Although France supports small populations of harbour and grey seals, both species are at the southern limit of their range so population information from this area is of particular interest.*'

WGMME have tasked themselves with making significant efforts in 2013 to update the seal database and further its inclusion in the ICES database.

7 Future work and recommendations

7.1 Future work of the WGMME

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

A list of the following recommendations can also be found at Annex 5 of this document.

Recommendation I

WGMME reiterates its **strong recommendation** that the bycatch management procedures developed under the SCANS-II and CODA projects (SCANS-II 2008; Winship, 2009; CODA, 2009) should be taken forward to develop management frameworks for bycatch at a European level. Without explicit conservation and management objectives, further development of bycatch management procedures is limited. It is proposed that WGMME and WGBYC collaborate to progress this approach as part of the MSFD indicator development during 2013.

Recommendation II

WGMME **strongly recommends** that Member States use the proposed management units for reporting requirements of the Habitats Directive and for the development of indicators and their assessment for the Marine Strategy Framework Directive. In summary, there is a single MU in European North Atlantic for common dolphin (*Delphinus delphis*), white beaked dolphin (*Lagenorhynchus albirostris*), white sided dolphin (*Lagenorhynchus acutus*) and minke whale (*Balaenoptera acutorostrata*). For bottlenose dolphin (*Tursiops truncatus*) there are ten separate units closely associated with the mainly resident inshore populations in the European North Atlantic and a separate MU for the wider ranging mainly offshore animals. For harbour porpoise (*Phocoena phocoena*), MUs are proposed for the Iberian Pennisula, Bay of Biscay, Celtic Sea (including SW Ireland, Irish Sea and Western Channel) and NW Ireland/West Scotland and the North Sea. The MUs for harbour porpoises may need to be revisited as indicators for MSFD become better defined and aligned with ICES rectangles to enable the calculation of more accurate bycatch estimates. For the purposes of MSFD, it maybe that consideration of the species will need occur at the regional seas level (e.g. North Sea).

Recommendation III

WGMME strongly supports the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2015 and **recommends** that it is supported by all range states and by ICES, ASCOBANS and the European Commission. Continuation of these surveys is essential for accurate population estimates, essential for reporting requirements of both the Habitats Directive and the Marine Strategy Framework Directive.

Recommendation IV

Following increasing understanding of harbour seal populations, WGMME reiterates its **recommendation** to OSPAR that the seal EcoQO subunits be updated. The revised harbour seal EcoQO should therefore read:

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 twelve sub-units of the North Sea. These sub-units are: Shetland; Orkney and north coast of Scotland; Moray Firth and East coast of Scotland; the Greater Wash/Scroby Sands; the French North Sea and Channel coasts; the Netherlands Delta area; the Wadden Sea; Heligoland; Limfjord; the Kattegat; the Skagerrak; the Oslofjord; and the west coast of Norway south of 62°N.

and for grey seals:

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. These subunits are: Orkney; Firth of Forth; the Farne Islands; the Greater Wash; the French North Sea and Channel coasts; the Wadden Sea; Heligoland; Kjørholmane (Rogaland).

Such a change in the subunits would more accurately reflect current monitoring and/or management areas. For the development of the MSFD indicators is recommended, however, that the subunits do not get specifically listed. Thus, avoiding the need to rewrite/update the wording of the indicator as new information on populations comes to light.

Recommendation V

WGMME **recommends** that power analyses are undertaken for all seal management areas to determine the trends in populations that can accurately be assessed with current monitoring practices. It is also recommended that ICG-COBAM give consideration to using the EcoQO approach rather than the more generic baseline proposed but that it would be useful to relate the percentage change to some earlier baseline such as the favourable reference population determined under the Habitats Directive.

Recommendation VI

It is **recommended** that WGMME assess the JCP outputs when they become available with a view to their contribution to international reporting requirements in 2013. The current Article 17 guidance for the 2013 reporting round includes a much greater emphasis on transboundary reporting where appropriate (European Commission, 2011). Further development/refinement of MSFD indicators of biodiversity will be required through 2013 and implementation of the monitoring needs to meet these requirements is needed by 2014. Development of an international equivalent of the JCP is also recommended that could be held by ICES.

Recommendation VII

The marine renewable industry is developing rapidly and regulators need to make decisions on granting consent for licensing in the near future. As the industry expands from a few sites to a large number of sites over larger areas of sea, it will become increasingly important to be able to predict population effects in order to meet management objectives such as Favourable Conservation Status under the Habitats Directive and Good Environmental Status under the Marine Strategy Framework Directive. A good management framework requires a sufficient level of basic understanding of animal-device interactions including a deploy and monitor strategy for assessing these interactions; it would also benefit from ongoing data collection (monitoring) at appropriate scales to allow the incorporation of a feedback mechanism and to enable determination of whether management actions are allowing objectives to be met. *The WGMME recommends the development of an appropriate precautionary management framework for marine renewable energy technologies.*

Annex 1: Participants list

Name	Address	Phone/Fax	E-mail
Steven Benjamins	The Scottish Association for Marine Science Dunstaffnage Marine Laboratory Oban PA37 1AQ Argyll Scotland UK	Tel: +44(0)1631-559449 (office) Fax: +44(0)1631-559001	Steven.Benjamins@sams.ac.uk
Andrew Brownlow Invited Expert	Scottish Agricultural College SAC Wildlife Unit IV2 4JZ Inverness United Kingdom	Phone +44 1463 246044 Fax +44	andrew.brownlow@sac.co.uk
Florence Caurant	University of La Rochelle-ILE LIENSS, UMR 7266 2 rue Olympe de Gouges 17 000 La Rochelle France	Phone +33 Fax+33	Florence.caurant@univ-lr.fr
Philip Hammond	Scottish Oceans Institute University of St Andrews East Sands KY16 8LB St Andrews Scotland United Kingdom		psh2@st-andrews.ac.uk
Anders Galatius Jørgensen	University of Aarhus Dept of Bioscience Frederiksborgvej 399 4000 Roskilde Denmark	Phone +45 871587021 Fax Cell: +45 28710372	agj@dmu.dk
Russell Leaper Invited Expert (by correspondance)	University of Aberdeen Canal House Banavie, Fort William PH 33 7LY United Kingdom	Phone +44 1397 772544 Fax +44	r.c.leaper@aber.ac.uk

Name	Address	Phone/Fax	E-mail
David Lusseau Invited Expert (by correspondance)	University of Aberdeen Institute of Biological and Environmental Sciences Zoology Building Tillydrone Avenue Room 415 Aberdeen AB24 2TZ, Scotland United Kingdom	Phone +44 1224 272843 Fax +44	d.lusseau@abdn.ac.uk
Veronique Lesage	Fisheries and Oceans Canada PO Box 100 850 Route de la Mer Mont-Joli QC G5H 3Z4 Canada	Phone +1 Fax +1	veronique.lesage@dfo-mpo.gc.ca
Klaus Lucke	IMARES Wageningen UR P.O. Box 167 1790 AD Den Burg Netherlands	Phone +31 317 487470 Fax +31 317 487362	klaus.lucke@wur.nl
Kelly Macleod By correspondence	Joint Nature Conservation Committee Inverdee House Baxter Street AB11 9QA Aberdeen United Kingdom	Phone +44 1224 266584 Fax Cell: 44 7881 504779	kelly.macleod@jncc.gov.uk
Sinéad Murphy	Institute of Zoology Zoological Society of London Regent’s Park NW1 4RY London United Kingdom	Phone +64 9 414 08000 ext. 41524	s.n.murphy@massey.ac.nz
Graham Pierce	University of Aberdeen Oceanlab Main Street AB41 6AA Newburgh Ellon, Aberdeenshire Scotland United Kingdom	Phone +44 1224 272459 Fax +44	g.j.pierce@abdn.ac.uk
Eunice Pinn Chair	Joint Nature Conservation Committee Inverdee House Baxter Street AB11 9QA Aberdeen Scotland United Kingdom	Phone +44 1224 266580	eunice.pinn@jncc.gov.uk

Name	Address	Phone/Fax	E-mail
Rui Conde de Araújo Brito Prieto da Silva By correspondence	University of the Azores Cais de Santa Cruz PT-9901 862 Horta Portugal	Phone +351 292 200 400 Fax +351 292 200411	rprieto@uac.pt
Vincent Ridoux	University of La Rochelle 23 avenue Albert Einstein 17071 La Rochelle Cedex 9 France	Phone +33 Fax +33	vincent.ridoux@univ-lr.fr
Marina Sequeira By correspondence	Institute for Conservation of Nature and Biodiversity Rua de Santa Marta, 55 1150-294 Lisbon Portugal	Phone +351 21 350 79 00 Fax +351 21 350 79 86	sequeiram@icnb.pt
Mónica Silva By correspondence	University of the Azores Centro do IMAR da Universidade dos Açores Cais de Santa Cruz PT-9901 862 Horta Portugal	Phone +351 292200442 Fax +351 292200411	monica@uac.pt
Raúl Castro Uranga By correspondence	AZTI-Tecnalia AZTI Pasaia Herrera Kaia Portualde z/g 20110 Pasaia (Gipuzkoa) Spain		rcastro@pas.azti.es
Gordon Waring By correspondence	National Marine Fisheries Services Northeast Fisheries Science Center Woods Hole Laboratory 166 Water Street Woods Hole MA 02536-1026 United States	Phone +1 508-495-2311 Fax +1 508-495-2258	gordon.waring@noaa.gov
Ben Wilson By correspondence	Scottish Association for Marine Science Dunstaffnage Marine Laboratory Oban PA37 1QA Argyll Scotland United Kingdom	Phone +44 1631 559346 Fax +44	ben.wilson@sams.ac.uk

Name	Address	Phone/Fax	E-mail
Matthew Witt Invited Expert By correspondence	University of Exeter College of Life and Environmental Sciences Cornwall Campus Treliever Road TR10 9EZ Penryn Cornwall United Kingdom	Phone +44 1326 371872 Fax +44	M.J.Witt@exeter.ac.uk

Annex 2: WGMME terms of reference for the next meeting

The **Working Group on Marine Mammal Ecology** (WGMME), chaired by Eunice Pinn, UK, will meet in Copenhagen, Denmark, 4–7 February 2013 to:

1) Review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals; specifically, the MUs for harbour porpoises will need to be revisited as indicators for MSFD become better defined. Such units will need to be aligned with the appropriate ICES rectangles to enable the calculation of more accurate bycatch estimates;

2) Collaborate with WGBYC to develop bycatch management procedures (based on the SCANS-II and CODA projects) for assessing bycatch at a European level. This work should include harbour porpoise (SCANS II), common dolphin (CODA) and consideration of additional species for which bycatch estimates have been made or suggested as a potential MSFD indicator. Such species include bottlenose dolphin, striped dolphin, harbour seal and grey seal;

3) 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 development of MSFD indicators, targets and appropriate baselines;

4) Update on development of database for seals and status of intersessional work, assessing its potential contribution to the development of MSFD indicators, targets and baselines;

5) Review and assess how the monitoring of effects around offshore wind and marine renewable energy devices is or could be undertaken.

WGMME will report for to the attention of the Advisory Committee.

Supporting Information

Priority	The European Commission guidance for the 2013 Article 17 reporting round has a much greater emphasis on transboundary reporting, where appropriate. All cetacean species in European waters will require such reporting. Tor 1 and 3 are therefore considered very high priority. WGMME is the only group that can support such a requirement. Member States are also being asked to develop indicators and targets for measuring good environmental status (as required by MSFD). It is essential that bycacth indicators, recognised as the most important anthropogenic threat to cetaceans, are fit for purpose. It is therefore also a high priority for WGBYC and WGMME to collaborate to further such work (Tor 2).
Scientific justification	Term of Reference 1) Development of approaiate management units/regions is esstential for the future development of MSFD indicators and targets, particularly those in relation to bycatch. Term of Reference 2) Work on the estimation of bycacth and its impact at the population level has been ongoing for many years. The use of a management framework approach has been reiterated consistently over the last four years by WGMME and ICES (2010) has also advised that the European Commission should move to such an approach. Term of Reference 3) WGMME (2009) reviewed the outputs of the first Article 17 reporting round and concluded that the focus on national waters meant that the collated reports for European North Atlantic waters did not reflect the status of the species accurately. It is essential that these reports are improved in future reporting rounds if they are to achieve the IUCN Red List status that European Commission aspires to. Term of Reference 4) little development of this database has occurred over the last two years. However the need for it has increased with the introduction of the MSFD. Term of Reference 5) The marine renewable indistry is developing rapidly. Over the last three years WGMME have review the potential impact of windfarms (2009), tidal devices (2010) and wave device (2011). By 2012, sufficient information should be available to enable a review of monitoring approaches.
Resource requirements	No specific requirements beyond the needs of members to prepare for, and participate in, the meeting.
Participants	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities	None.
Financial	No financial implications.
Linkages to advisory committees	WGMME reports to ACOM.
Linkages to other committees or groups	SCICOM, SSGSUE
Linkages to other organizations	The work undertaken with respect to the MSFD builds on and links with that of OSPAR's ICG-COBAM group. ASCOBANS and ACCOBAMS have also proposed joint working groups looking at marine renewables and development of indicators for MSFD.

Annex 3: Proposed indicators from marine mammal strandings

Criterion: Population condition (1.3)

Indicator: Mortality rates (1.3.1.)

Parameter/metric: Numbers of individuals within species whose death was caused by anthropogenic activity based on recovered stranded animals examined at necropsy.

Draft: 10.05.2012.

Background

Strandings monitoring involves the opportunistic collection of data from animals found on shore or submitted as bycatch to collect information of the causes of mortality, on health status, disease, contaminants, life history and diet. Examination of a stranded marine animal is the only way of accurately establishing a cause of death for that individual. Data on population dynamics (age-structure) and the life history (pregnancy rates, diet) of wide-ranging marine mammal species have been primarily ascertained via analysis of necropsy samples. Analysis of samples collected at necropsy is required for investigation of complex ecological interactions, for example the relationship between contaminant burdens, health status and reproductive success (Jepson *et al.*, 2005). Stranded animals are a subset of the at sea population and are biased towards sick animals, coastal animals and species and individuals which remain buoyant when dead. Conversely, monitoring stranded animals is an efficient and cost effective way enhancing our knowledge of species biology and provides a useful sentinel function to changing conditions and new threats.

Criterion	Indicator	Parameter/Metric	Target
Population Condition (1.3)	Mortality rates (1.3.1.)	Numbers of individuals within species whose death was caused by anthropogenic activity based on recovered stranded animals examined at necropsy	The annual anthropogenic mortality rate of [marine mammal species] is reduced below a threshold of [X% of necropsies]

Parameters/metrics

Stranding schemes have been developed in most European countries for many decades, in part due to the ASCOBANS requirement for such work. They therefore offer a unique large scale and long-lasting tool for the continuous surveillance of causes of mortality (both natural and anthropogenic) in marine mammal populations. Adequate monitoring of marine mammal strandings requires good organization and a long term commitment of resources, normally through coordinated networks. Recovery and necropsy of cases and subsequent analyses of samples require a minimum of infrastructure and investment to ensure sufficient confidence in the validity of the results. Extrapolating the results of the individual to the state of the population has known biases and sufficient number of cases are required for that process to be scientifically robust. Data recovered by evaluation of strandings are however fundamental to amassing information at a fine enough resolution to effectively evaluate many current and emerging threats to populations.

Current state of implementation/current monitoring

Necropsy indicators are designed to identify the impact of a predefined list of activities and pressures, natural and anthropogenic, which affect the fitness of the individual and, by extrapolation, the ecosystem. Whilst it not always possible to infer causality from observed pathology, collection of a suite of indicators offers a potentially powerful way of assessing complex and multifactor stressors on individuals and populations. This is particularly important in degraded or stressed ecosystems where additional stressors may have multiplicative or other non-linear effects.

Indicators establishing trends in the anthropogenic impact on cetacean populations over time require certain key assumptions are met. For any given surveillance area, animals should undergo standardized and harmonised necropsies which ensure good observer agreement between cases and confidence in the accuracy of the diagnosis. The cause of death indicator is defined as the primary reason the animal became stranded, categorised into class (three categories) and further subcategorised, if appropriate, into specific cause (Table 1). Categorical cause of death can thus be used to calculate absolute and proportional causes of mortality rates, e.g. of bycatch mortality etc. based on proportion of carcases with diagnosed causes of death which are attributable to bycatch. The cause of death can be aggregated to primarily anthropogenic and non-anthropogenic causes.

Table 1. Example categories for cause of death as established by necropsy (derived from current UK protocols).

Cause of death class	Cause of death subcategory	Primary anthropogenic cause?
Trauma	Bycatch	Yes
	Bycatch (known)	Yes
	Cold Stunned	No
	Dystocia & Stillborn	No
	Entanglement	Yes
	Physical Trauma (non specific)	No
	Physical Trauma Spiral lacerations	Yes
	Physical Trauma Boat/Ship Strike	Yes
	Physical Trauma Bottlenose Dolphin Attack	No
Infectious Disease	(Meningo)encephalitis	No
	Gastritis and/or Enteritis	No
	Generalised Bacterial Infection	No
	Generalised Mycotic Infection	No
	Others	No
	Pneumonia Bacterial	No
	Pneumonia Bacterial and Mycotic	No
	Pneumonia Mycotic	No
	Pneumonia Parasitic	No
	Pneumonia Parasitic and Bacterial	No
	Pneumonia Parasitic and Mycotic	No
	Pneumonia Unknown Aetiology	No
Others	Gas Embolism	Unknown
	Live Stranding	Unknown
	Neonatal Death	No
	Neoplasia	No
	Not Established	Unknown
	Others	Unknown
	Pneumonia Unknown Aetiology	No
	Starvation	No
	Starvation (neonate)	No

The sample requirements for this variety of indicators can be scaled to meet current national monitoring and surveillance programmes, although, a commitment to undertake a minimum of monitoring and analytical work will be required. National systems to collect data from strandings have been in operation in some countries for decades and a large body of historical data already exists. Harmonisation of protocols and samples and a system for rapid exchange of information on new strandings is recommended however to 'add value' to the information collected by individual Member States. This would allow for a ecosystem-wide surveillance system, providing a more rapid alert to transboundary mortality events and to share data to increase power of statistical analysis on species with a low incidence of stranding.

Target setting

Changes in the various causes of death are likely to be indicative of changing anthropogenic influences where those causes of death are associated with human activity. It is particularly important to minimize biases arising from differential capacity to determining death cause as a result of carcass decomposition condition, observer training and available necropsy facility. The careful definition of a standard baseline examination protocol will need to be agreed between stranding schemes to ensure large spatial coverage, large sample size and robust conclusions. To have confidence in both these point prevealances and any trends observed over time requires examination of a sufficient number of cases based on *a priori* power analysis.

Further work will be required to identify the most appropriate threshold for the target and the number of necropsies required on an annual basis (by species) to identify trends in the mortality rate.

Future steps necessary

There is clearly a lack of information on aspects of this indicator, particular the most appropriate threshold for assessment of GES.

1) Harmonisation of individual Member States pathology protocols for conducting necropsies, storing samples and conducting contaminant analyses within European waters would be beneficial (ICES WGMME 2010).

2) Development of a proportional incidence rate for cause of death for agreed number of sentinel species (e.g. harbour porpoise, common dolphin, minke whale, white beaked dolphins, long finned pilot whale).

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

4) Development of an online database for data viewing.

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Proposed indicators from marine mammal strandings continued

Criterion: Population condition (1.3).

Indicator: Mortality rates (1.3.1.).

Parameter/metric: Blubber PCB toxicity threshold concentration of 13 mg/kg lipid wt (summed ICES7 congeners) for marine mammals.

Draft: 04/05/2012.

Background

Marine mammals are exposed to a range of potentially toxic chemicals in their environment as some lipophilic and persistent organic compounds bioaccumulate to very high levels, particularly in top predators such as harbour porpoises, bottlenose dolphins and killer whales. Detailed research on UK-stranded harbour porpoises conducted under the UK Cetacean Strandings Investigation Programme (www.ukstrandings.org) has shown strong links between elevated blubber PCB levels and infectious disease mortality (Jepson *et al*., 1999; Jepson *et al*., 2005; Hall *et al*., 2006) consistent with fatal PCB-induced immunosuppression.

In one case-control study of UK-stranded harbour porpoise the risk of infectious disease mortality increases 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. In a second case-control study of UK-stranded harbour porpoises, mean summed 25CB congeners in the "healthy" control group (death due to physical trauma) was 13.6 mg/kg lipid and 27.6 mg/kg lipid for the animals that died of infectious diseases (Jepson *et al*., 2005). These studies allow an estimated threshold of toxicity (including immunosuppression and reproductive impairment) for blubber PCB concentrations that exceed 20 mg/kg lipid wt (for summed 25CB congeners) in harbour porpoises. This equates to a blubber PCB toxicity threshold concentration of 13 mg/kg lipid wt (for summed ICES7 CB congeners) based on standard regressions between summed 25CBs and summed ICES7 CB congeners. This 13 mg/kg concentration for summed ICES7 congeners could be used for other marine mammal species (not just harbour porpoises) to assess populations that may at risk toxic effects at individual and population levels.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/Metric	Target
Population condition	Mortality rate	Blubber PCB toxicity threshold concentration of 13 mg/kg lipid wt for marine mammals (for summed ICES7 CB congeners)	Biological effects from contaminants are kept within safe limits, so that there are no significant impacts on, or risks to, marine mammals

Parameters/metrics

This is a European wide indicator. 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.

Analytical methods for PCB concentrations in tissues are both highly sensitive and internationally standardised for comparison with tissue PCB levels in other regions. The use of an established threshold concentration for PCB-induced toxicity using empirical cetacean data enables the early detection and assessment of cetacean individuals and populations exposed to levels of PCBs that are likely to induce potentially lethal toxic effects including immunosuppression and reproductive impairment. This indicator is calculated based on measuring the ICES7 CB congeners (52, 101, 118, 138, 153, and 180). As female cetaceans can offload the majority of their PCB burden to their first born offspring during pregnancy and lactation, data on age and reproductive status (i.e. sexually immature, pregnant, lactating, resting) and whether the individual was previously gravid, have to be assessed in order to provide context to the estimated contaminant burden. For males, age is the most important criteria.

Current state of implementation/current monitoring

PCBs have the potential to cause death and impair reproduction in populations with highest exposure. High PCB exposure also has the potential to inhibit the recovery of depleted populations historically exposed to other forms of anthropogenic pressures (e.g. hunting/bycatch). Although management interventions to directly reduce PCB exposure are rather limited for cetaceans, targeted/prioritised conservation measures to limit all other anthropogenic pressures (e.g. bycatch) may be required in those species with highest PCB exposure (e.g. *Tursiops truncatus* and *Orcinus orca*). As part of the various European cetacean stranding programmes, cause of death, health status and nutritional condition of individuals are investigated. Blubber samples are collected, wrapped in foil and stored frozen (-20°C) for subsequent toxicological analysis. 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.

Analysis undertaken during the EC-funded 5th Framework BIOCET project (2001–2003) revealed that PCB concentrations in female harbour porpoises exceeded the threshold of 17 mg/kg (Kannan *et al.*, 2000) in almost three-quarters of the southern North Sea sample and over one-third of the Scottish sample (Pierce *et al.*, 2008). Whereas 40% of sampled common dolphins (sampled in Scotland, Ireland, France and Spain) exceeded the PCB threshold level (Pierce *et al.*, 2008). As part of a subsequent study, all sampled immature 'healthy' female common dolphins (1992–2006, n = 20) in English and Welsh waters had PCB levels above the threshold level of 17 mg/kg for adverse health effects, in addition to 59% of immature female harbour porpoises (Murphy 2009; Murphy *et al.*, 2010; Murphy *et al.*, 2012). The differing results between the two species may reflect differing life-history traits, e.g. common dolphins attain sexually maturity at a much older age than harbour porpoises (8.23 years vs. 4.51 years), which lengthens the period for accumulation of contaminants through dietary input during the immature phase (Murphy, 2009). Management interventions to reduce exposure to PCBs at both the national and international level have already been relatively effective and the levels recorded in cetaceans are declining, albeit slowly. In UK harbour porpoises, an initial decline in blubber Σ25CBs concentrations was observed in the mid-1990s, but this then plateaued off from 1997–present indicating that toxic impacts of PCBs will continue for some time (Law *et al.*, 2010). There is considerable concern that PCBs will accumulate to greater levels in marine mammals and other biota living in enclosed oceanic regions (like the Baltic and Mediterranean Seas; ICES 2010). PCB accumulation will also be increased by biological effects such as trophic level in some top predator species like inshore bottlenose dolphins and killler whales have some of the very highest PCBs levels every recorded on earth (Deaville and Jepson, 2011; ICES, 2010).

Target setting

It is recommended that the biological effects from contaminants are kept within safe limits, so that there are no significant impacts on, or risks to, marine mammals. The cause-and-effect relationships need to be established and monitored, as well as the impacts of accumulated (independent and interactive) effects. In order to undertake these tasks, knowledge of information on population growth rates, population structure, (life-history) biological parameters and density-dependent changes in these parameters are required.

Future steps necessary

1) Ensure there is a standardized sample and data collection protocol for stranded animals and biopsy of free-living cetaceans in European waters.
2) Assess PCB exposure in species/populations in Europe (e.g. harbour porpoise, killer whales, and bottlenose dolphins) using necropsy of stranded animals and biopsy of free-living animals.
3) Assessment of current population/management unit levels of relevant species with greatest exposures. Assess whether inshore and offshore bottlenose dolphin ecotypes exist in Europe and, if they do, whether they accumulate different PCB exposures and require different management and mitigation strategies.
4) Develop dose-response relationships between PCBs and health impacts (e.g. increased susceptibility to infectious disease mortality and reproductive impairment) for cetacean populations with consistently high PCB exposure (above proposed blubber PCB toxicity thresholds). Quantify the risk of high PCB exposure to individuals and populations.

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Proposed indicators from marine mammal strandings continued

Criterion: Population condition (1.3).

Indicator: Population demographic characteristics (e.g. body size or age-class structure, sex ratio, fecundity rates, survival/mortality rates).

Parameter/metric: Assessing temporal changes in population pregnancy rates, proportion of mature individuals, proportion of females simultaneously pregnant and lactating, average age attained at sexual maturity, nutritional condition, and variations in reproductive parameters with age.

Draft: 04/05/2012.

Background

Estimates of reproductive parameters in marine mammals can be used to assess changes in dynamics of populations as a result of incidental bycatch and/or climate change. Further, they allow assessment of the long-term effects from anthropogenic toxins, such as PCBs and DDT, and infectious disease outbreaks on reproductive out at the individual and population level. Although evidence is extremely limited at this time, it is thought that anthropogenic noise may also affect reproductive rates (Wright *et al*., 2009).

This indicator responds to three main anthropogenic activities that alter population densities: incidental capture (bycatch), pollutants and reduced prey availability due to over fishing (or non-anthropogenic activities). However, in order to interoperate reproductive data correctly, population abundance estimates, trends in abundance and data on parameters that affect the dynamics of the population, such as annual mortality rates in fisheries, temporal variations in prey abundance and levels of anthropogenic toxins are required. It also requires an assessment of the cause of death, health status, nutritional condition, and the status of reproduction tract and organs during post mortem examinations. Age determination using teeth samples and gross and histological examination of gonadal material are undertaken during subsequent analysis. Once data are available, estimating population pregnancy rates and other reproductive parameters is relatively straight forward.

Temporal variations in the reproductive parameters can occur due to alterations in the availability of prey resources and population density. Cetacean populations are regulated through density-dependent changes in reproduction and survival, and it has been proposed that food resources are the main causative agent in the expression of density dependence, resulting in an increase in population growth rates (and reproductive output) at low densities (e.g. following large scale incidental mortality in fishing gear) and a decrease in growth rates (and reproductive output) at high densities. However, anthropogenic toxins and disease can alter reproductive rates by decreasing fertility, and causing abortions, premature parturition and neonatal mortality.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/Metric	Target
Population Conition (1.3)	Population demographic characteristics (e.g. body size or age-class structure, sex ratio, fecundity rates, survival/mortality rates)	Assessing temporal changes in population pregnancy rates, proportion of mature individuals, proportion of females simultaneously pregnant and lactating, average age attained at sexual maturity, nutritional condition, and variations in reproductive parameters with age	No statistically significant deviation from long-term variation

Parameters/metrics

Reproductive parameters such as population pregnancy rates, proportion of mature individuals, proportion of females simultaneously pregnant and lactating, average age attained at sexual maturity, and variations in reproductive parameters with age, can be used as an indicator of a change in population growth rates.

For monitoring, reproductive and condition parameters will be estimated for the Northeast Atlantic populations of the common dolphin and harbour porpoise, as well as the Iberian harbour porpoise population. Where sufficient data are available on a 5–10 yearly basis reproductive parameters will be estimated for populations of other small delphinids e.g. striped dolphins, bottlenose dolphins, white-beaked and white-sided dolphins.

Although the pregnancy rate is easily measured, power analysis suggested that extremely large variations in the common dolphin pregnancy rate of the Northeast Atlantic population would have to occur, in order to detect a statistically significant increase or decrease in the pregnancy rate (Murphy *et al.*, 2009). At a power of ≥80%, and an initial pregnancy rate of 25%, a sample size of >150 mature females would be required to detect an absolute decline of >13% in the pregnancy rate, whereas a sample size of >100 mature females would detect a decline >16%. A sample size of 50 mature females however, would only detect a decline of >20% (pregnancy rate at 0.05 or below) and at a lower power of 72%. In contrast, if an increase occurred in the pregnancy rate, a sample size of >150 mature females would be needed to detect a >16% increase in the pregnancy rate at a power of ≥80%. Adequate age and reproductive data from males and females (at least 50 individuals of each sex) are vital for estimating the average age attained at sexual maturity. Obtaining such a large sample size of sexually immature and mature individuals is difficult, and requires that European stranding and observer bycatch programmes continue sampling all available and suitable carcasses. One compromise would be to alter the criteria used for significance. Many managers remain unaware that the standard criteria usually used for significance (i.e. the risk of a type 1 error occurring; α=0.05) is not an objective scientific value but a policy choice based on the most commonly used level of statistical significance (Taylor and Gerrodette, 1993).

Knowledge of extrinsic factors such as bycatch rates and contaminant loads are required to give context to cross-sectional life-history information.

Current state of implementation/current monitoring

As part of the various European cetacean stranding programmes, cause of death, health status and nutritional condition of individuals are investigated. Teeth, ovaries and testes are collected for subsequent analysis assessing reproductive parameters such as maturity status and age.

The pregnancy rate has recently been estimated for the Northeast Atlantic common dolphin population (Portugal to Scotland; Murphy *et al*., 2009), and various regions within the harbour porpoise continuous system Northeast Atlantic population (Murphy, 2008; Winship, 2008; Learmonth *et al*., in review). Stranded common dolphins do provide a representative sample of the NE Atlantic population for estimating biological parameters (Murphy *et al*., 2009). Primarily as a large proportion of individual were incidentally captured in fishing nets and subsequently washed ashore, and older individuals (>20 years) were not over-represented in the stranding and bycatch data (Murphy, 2004; Jepson, 2005; Murphy *et al*., 2009). In addition there is a lower incidence of disease in stranded common dolphins compared to stranded porpoises (Jepson, 2005). However, due to the large proportion of diseased porpoises that strand in European waters on an annual basis, pregnancy rates should be estimated from animals that died from trauma, e.g. incidental capture, in order to obtain a representative sample of the wider population.

Combining samples/data from all European countries within the range of Northeast Atlantic common dolphin and harbour porpoise populations would allow an assessment of the reproductive parameter indicator on a five year basis for both these species. For the Iberian harbour porpoise population, this indicator would necessitate collaboration between Spain and Portugal, and due to its small size an assessment of reproductive parameter indicator on a ten year basis.

For other small delphinids, such as striped dolphins, bottlenose dolphins, white-sided and white-beaked dolphins, reproductive parameters as yet have to be determined. Again, this necessitates the involvement of all European countries within range of the respective populations in order to obtain sufficient sample sizes for analysis.

Target setting

No statistically significant deviation from long-term variation. Though it should be noted changes may become biologically significant before they can be detected statistically. Due to the natural life history of cetaceans and the need for sufficient samples, reproductive parameters are likely to be reestimated every five-to-ten years rather than annually.

Where the power of a monitoring scheme (e.g. >80%, $\beta = 0.2$) to detect change is different from the level of significance (e.g. 0.05) there is an imbalance in the risks of under and over protection. For the common dolphin, using a lower significance level of 0.2, a power of ≥80%, and an initial pregnancy rate of 25%, a sample size of only 50 mature females would be required to detect an absolute decline of >16% in the pregnancy rate, and an absolute increase of >20% in the pregnancy rate.

Assessments on required sample size, and level of detection would need to be estimated for other small delphinids and the harbour porpoise.

Future steps necessary

1) Development of a baseline for each population/management unit (where possible).
2) Assessment of current population/management unit reproductive parameters in small delphinids (already assessed for the common dolphin) and for the Northeast Atlantic harbour porpoise.
3) Development of a standardized sample and data collection protocol.
4) Development of an assessment tool for all species.

Literature

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

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

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Annex 4: Draft OSPAR Common Indicators

Marine mammals, number 31 and 33

Criterion: Species distribution (1.1.).

Indicator: Distributional range (1.1.1.) and distributional pattern within range (1.1.2.).

Parameter/metric: Distributional range and pattern of grey and harbour seal haul-outs and breeding colonies (HD).

Draft: 24.02.2012.

Background

Marine mammals, including harbour and grey seals, are top predators, and comprise an important part of biodiversity (D1). Harbour and grey seals have also been suggested as suitable indicators for a good environmental status of the foodweb as key trophic species (D4), but as they are mostly opportunistic feeders, and their diet varies by area, this might not be the case. As harbour and grey seal are taken up under the Habitats Directive (Annex II), their distribution comprises a key aspect for securing and achieving GES according to the MSFD.

It is generally possible to detect deterioration or improvement of the distribution of harbour and grey seal by monitoring their presence on existing (and former) breeding colonies or haul-out sites. When recording changes, it is necessary to assess and interpret these, in order to discriminate natural vs. human-induced causes, as there is no direct cause–effect relationship for change put forward. Changes due to climate change and 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.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/ Metric	Target
Species distribution	Distributional range	Distributional range of grey and harbour seal haul-outs and breeding colonies	No decrease with regard to baseline due to anthropogenic activities

Parameters/metrics

Existing OSPAR EcoQO's deal with grey seal pup production and population size of hauled-out harbour seals, but there is clearly an overlap with the distributional range, and an overlap between range and distributional pattern. The same monitoring will be used to undertake both analyses. For monitoring the EcoQO's on seals, the North Sea has been subdivided into different management units (respectively nine and eleven for grey and harbour seal). A subdivision into management units should be made for the whole range of both species, with indications of current and former occupancy.

Current state of implementation/current monitoring

There is sufficient monitoring at seal haul-out sites and at colonies. This monitoring takes place in combination with the monitoring of the parameters 33 (distributional pattern), 35 (abundance) and 37 (pup production).

Target setting

The proposed target is "no decrease with regard to the baseline due to anthropogenic activities". Some difficulties can be encountered here, because there is usually no straightforward link between the parameter and human activities. Although the baseline should be based on historical data, these are not available everywhere. Moreover, the historical distributional range of haul-out sites and colonies is a situation that cannot realistically be restored, given for instance coastal developments and tourism, and climatic changes may have important consequences. It is therefore likely that a modern baseline will have to be utilized.

Future steps necessary

Future steps are similar for the parameters 33 (distributional pattern), 35 (abundance) and 37 (pup production).

1) Compilation of existing data on the distributional range.

2) Subdivision of the area (beyond the North Sea) into management units. This exists already in the UK.

3) Development of a baseline for each management unit.

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

5) Development of an assessment tool.

Literature

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

Marine mammals, numbers 32 and 34

Criterion: Species distribution (1.1).

Indicator: Distributional range (1.1.1.) and distributional pattern with range (1.1.2).

Parameter/metric: Distributional range and pattern of cetaceans species regularly present.

Draft: 01.03.2012.

Background

Marine mammals, including cetaceans, are top predators, and comprise an important part of biodiversity (D1). Cetaceans have also been suggested as suitable indicators for a good environmental status of the foodweb as key trophic species (D4). However, for the more abundant species in European waters, the diet varies spatially and temporally, so this might not be the case. As all cetacean species are taken up under the Habitats Directive (Annex IV), their distribution comprises a key aspect for securing and achieving GES according to the MSFD.

The distribution of cetaceans can be monitored using a variety of techniques (e.g. visual surveys from vessels and planes and towed hydrophone arrays). With the exception of 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 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.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/Metric	Target
Species distribution	Distributional range (1.1.1) and Distributional pattern within range (1.1.2)	Distributional range of cetacean species regularly present (32) and Distributional pattern at the relevant temporal scale of cetacean species regularly present (34)	No decrease with regard to baseline due to anthropogenic activities

Parameters/metrics

There is a very clear overlap between range and distributional pattern with 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 FCS assessments for the Habitats Directive. 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).

Current state of implementation/current monitoring

Monitoring is undertaken through a variety of routes and organisations. There are large scale international surveys such as SCANS and CODA that occur at a decadal scale, annual national surveys occur in the waters of some Member States and, at a more localised scale, there are various surveys undertaken by the state, academic institutions and/or non-governmental organisations.

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; Paxman and Thomas, 2010).

The monitoring and assessment undertaken for parameters 32 and 34, will be in combination with parameter 36 (abundance).

Target setting

The proposed target is "no decrease with regard to the baseline due to anthropogenic activities". Some difficulties can be encountered here, because there is usually no straightforward link between parameters 32/34 and human activities. 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 (i.e. pre-commercial hunting) 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. It is likely that a modern baseline will have to be utilised, such as that provided through the JCP analyses.

Future steps necessary

Future steps are similar for the parameters 32 (distributional range), 34 (distributional pattern) and 36 (abundance).

1) Compilation of existing data on the distributional range. This has already begun through the JCP, but it is recognised that a number of significant national datasets are missing.
2) Development of a baseline for each species.

3) Development of a standardized monitoring methodology, or alternatively a mechanism for standardizing data post collection. Effort-related monitoring of cetaceans is to some extent standardised. It is the standardising of data post collection that is necessary, although this has already been started through the JCP.

4) Development of an assessment tool.

Literature

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Marine mammals, number 35

Criterion: Population Size (1.2).

Indicator: Species: Population abundance and/or biomass, as appropriate (1.2.1).

Parameter/metric: Abundance of grey and harbour seal at haul-out sites & within breeding colonies.

Draft: 01.03.2012.

Background

Marine mammals, including harbour and grey seals, are top predators, and comprise an important part of biodiversity (D1). Harbour and grey seals have also been suggested as suitable indicators for a good environmental status of the foodweb as key trophic species (D4), but as they are mostly opportunistic feeders, and their diet varies by area, this might not be the case. 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.

It is possible to detect changes in abundance of harbour seals from haul-out counts and for grey seals from pup counts. When recording changes, it is necessary to assess and interpret these, in order to discriminate natural vs. human-induced causes, as there is no direct cause–effect relationship for change put forward. Changes due to climate change and 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.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/Metric	Target
Populatioon size	Species: Population abundance and/or biomass, as appropriate	Abundance of grey and harbour seal at haul-out sites & within breeding colonies	No statistically significant decrease with regard to baseline due to anthropogenic activities

Parameters/metrics

Existing OSPAR EcoQO's deal with 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 there is clearly an overlap with the parameters distributional range (31) and distributional pattern with range (33). For monitoring the EcoQO's on seals, the North Sea has been subdivided into different management units (respectively nine and eleven for grey and harbour seal). A subdivision into management units should be made for the whole range of both species, with indications of current and former abundance.

Current state of implementation/current monitoring

There is sufficient monitoring at seal haul-out sites and at colonies. This monitoring takes place in combination with the monitoring of the parameters 31 (distributional range), 33 (distributional pattern) and 37 (pup production).

Target setting

The proposed target is "no statistically significant decrease with regard to baseline due to anthropogenic activities". Some difficulties can be encountered here, because there is usually no straightforward link between the parameter and human activities. Although the baseline should be based on historical data, these are not available everywhere. Moreover, it may not be possible to restore harbour seals to historical abundance levels because of large-scale coastal development, tourism, and the impacts of epizooitics such as PDV in combination with recent increases in the grey seal population. Climatic changes may have important consequences for both species.

Future steps necessary

Future steps are similar for the parameters 31 (distributional range), 33 (distributional pattern) and 37 (pup production).

1) Compilation of existing data on abundance of both species.
2) Subdivision of the area (beyond the North Sea) into management units (although management areas already exist in the UK).
3) Development of a baseline for each management unit.
4) Development of a standardized monitoring methodology, or alternatively a mechanism for standardizing data post collection.
5) Development of an assessment tool.

Literature

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

Marine mammals, number 36

Criterion: Population size (1.2).

Indicator: Population abundance and/or biomass, as appropriate (1.2.1).

Parameter/metric: Abundance at the relevant temporal scale of cetacean species regularly present.

Draft: 01.03.2012.

Background

Marine mammals, including cetaceans, are top predators, and comprise an important part of biodiversity (D1). Cetaceans have also been suggested as suitable indicators for a good environmental status of the foodweb as key trophic species (D4). However, for the more abundant species in European waters, the diet varies spatially and temporally, so this might not be the case. As all cetacean species are taken up under the Habitats Directive (Annex IV), their abundance comprises a key aspect for securing and achieving GES according to the MSFD.

The abundance of cetaceans can be monitored using a variety of techniques (e.g. visual surveys from vessels and planes and towed hydrophone arrays). 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 decline or increase in population size 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.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/Metric	Target
Population Size	Population abundance and/or biomass, as appropriate (1.2.1)	Abundance at the relevant temporal scale of cetacean species regularly present	No statistically significant decrease with regard to baseline due to anthropogenic activities

Parameters/metrics

The same monitoring used to assess changes in cetacean abundance will be used to assess changes in distribution. An assessment of abundance, including trends over time, is required as part of the FCS assessments for the Habitats Directive. 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).

Current state of implementation/current monitoring

Monitoring is undertaken through a variety of routes and organisations. There are large scale international surveys such as SCANS and CODA that occur at a decadal scale, annual national surveys occur in the waters of some Member States and, at a more localised scale, there are various surveys undertaken by the state, academic institutions and/or non-governmental organisations.

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; Paxman and Thomas, 2010).

The monitoring and assessment undertaken for parameter 36, will be in combination with that for parameters 32 (distributional range) and 34 (distributional pattern within range).

Target setting

The proposed target is "no statistically significant decrease with regard to baseline due to anthropogenic activities". Some difficulties can be encountered here, because there is usually no straightforward link between parameter 36 and human activities. Although the baseline should be 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. Climatic changes may have important consequences. It is likely that a modern baseline will have to be utilised, such as that provided through the SCANS/CODA surveys.

Future steps necessary

Future steps are similar for the parameters 32 (distributional range), 34 (distributional pattern) and 36 (abundance).

1) Compilation of existing data on abundance. This has already begun through the JCP, but it is recognised that a number of significant national datasets are missing.
2) Development of a baseline for each species.
3) Development of a standardized monitoring methodology, or alternatively a mechanism for standardizing data post collection. Effort-related monitoring of cetaceans is to some extent standardised. It is the standardising of

data post collection that is necessary, although this has already been started through the JCP.

4) Development of an assessment tool.

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.

Marine mammals, number 37

Criterion: Population Condition (1.3).

Indicator: Population demographic characteristics (e.g. body size or age-class structure, sex ratio, fecundity rates, survival/mortality rates) (1.3.1).

Parameter/metric: Harbour seal and Grey seal pup production.

Draft: 01.03.2012.

Background

Marine mammals, including harbour and grey seals, are top predators, and comprise an important part of biodiversity (D1). Harbour and grey seals have also been suggested as suitable indicators for a good environmental status of the foodweb as key trophic species (D4), but as they are mostly opportunistic feeders, and their diet varies by area, this might not be the case. 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 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 newborn 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 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 area (notably the Wadden Sea), counts are made during the breeding season for harbour seals.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/Metric	Target
Population condition	Population demographic characteristics (e.g. body size or age-class structure, sex ratio, fecundity rates, survival/mortality rates)	Harbour seal and Grey seal pup production	No statistically significant deviation from long-term variation/no decline of ≥10%

Parameters/metrics

For monitoring, the North Sea has been subdivided into different management units (respectively nine and eleven for grey and harbour seal). A subdivision into management units should be made for the whole range of both species. An existing

OSPAR EcoQO deals with grey seal pup production. There is not an equivalent to harbour seal pup production.

Population condition monitoring for grey seal pup production clearly overlaps with that undertaken for range, distributional pattern with range and abundance. The same monitoring will be used to undertake all these analyses.

For harbour seals the situation is slightly different. Harbour seal counts are undertaken during the breeding season in the Wadden Sea, but in the UK there are only two management units for which pup counts can be undertaken (Moray Firth, Scotland, and the Wash, England). These data are currently held as counts.

Current state of implementation/current monitoring

There is sufficient monitoring at breeding colonies for grey seals. In contrast, for harbour seals it will not be possible to cover all management units. The monitoring required takes place in combination with the monitoring for the parameters 31 (distributional range), 33 (distributional pattern) and 35 (abundance).

Target setting

The proposed target is "No statistically significant deviation from long-term variation/no decline of ≥10%". Some difficulties can be encountered here, because there is usually no straightforward link between the parameter and human activities. Although the baseline should be based on 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, and climatic changes may have important consequences. It is therefore likely that a modern baseline will have to be utilized.

Future steps necessary

Future steps are similar for the parameters 31 (distributional range), 33 (distributional pattern) and 35 (abundance).

1) Compilation of existing data on pup counts and production estimates.
2) Subdivision of the area (beyond the North Sea) into management units (already exists for the UK). Assessment in all management units will not be possible for harbour seal.
3) Development of a baseline for each management unit (where possible).
4) Development of a standardized monitoring methodology, or alternatively a mechanism for standardizing data post collection.
5) Development of an assessment tool.

Literature

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

Marine mammals, number 38, 39

Criterion: Population condition (1.3).

Indicator: Mortality rates (1.3.1).

Parameter/metric: Numbers of individuals within species being bycaught in relation to population.

Draft: 01.03.2012.

Background

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 35 and 36).

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 can also be set on pinnipeds, as bycatch also occurs in these marine mammals. As the maximum population growth rates differ in marine mammals, different targets (in % of the population) will be needed. Given the high mobility of marine mammals, and the distributional range of populations, assessments will necessarily be made on a wide scale (population range). Difficulties exist in both measuring bycatch and population size in a sufficiently high accuracy to draw conclusions, and in combining data originating from different regions for an overall assessment of GES.

Indicators with suggested parameter and target.

Criterion	Indicator	Parameter/Metric	Target
Population condition	Species population demographic characteristics (mortality rate)	Numbers of individuals being bycaught in relation to population estimates	The annual bycatch rate of [marine mammal species] is reduced to less than [X]% of the best population estimate

The target will depend on the species. Obvious species for which the target could/should be set, as bycatch exists, are harbour porpoise, harbour seal, grey seal, short beaked common dolphin, bottlenose dolphin and striped dolphin. Regional differences in the species selected exist.

Parameters/metrics

The target commonly accepted for harbour porpoises (existing OSPAR EcoQO, ASCOBANS resolution, IWC) is less than 1.7% of the best population estimate. For other species, such as the common dolphin, the population against which the target should be set is less straightforward. For seals it may be set against a measure of the regional population size, as there is a fairly good knowledge of the number of seals at haul-out sites and breeding colonies. Good information might also be available for coastal bottlenose dolphins.

An alternative for the parameter is the use of the current bycatch rate as the baseline. This would mean that no information is required on the population size, but have the disadvantage that there is no link with the population state.

Current state of implementation/current monitoring

The 1.7% limit for the harbour porpoise is widely accepted, and should be implemented by at least the states which are a member of ASCOBANS. There is a requirement for monitoring bycatch of cetaceans in fisheries legislation (e.g. Regulation 812/2004; Data Collection Regulation) and in the Habitats Directive. Monitoring of marine mammal populations is diverse, with parts of some populations being regularly monitored on a regional scale (e.g. seals at colonies) and with other populations only monitored in approximately decadal large scale surveys (e.g. SCANS surveys for harbour porpoises and common dolphins), yielding population estimates for one season only.

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 (ICES, 2008b). It was quickly apparent that many of the fisheries suspected to have the highest bycatch levels are conducted without bycatch observer programmes as these are not a requirement of Council Regulation 812/2004. Subsequently, ICES Working Group on Bycatch of Protected Species has tried annually to evaluate the impact of fisheries bycatch.

Extrapolated estimates of total bycatch in 2009 were available for striped dolphins (about 870), for common dolphins (around 1500), for bottlenose dolphins (ten) and for harbour porpoises (about 1100; ICES 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 (ICES, 2011). As a consequence, it is not currently possible to evaluate whether such an indicator will provide an accurate assessment of GES, but 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 fishery sector, which is considerably harder to monitor. An alternative might be for the target to be reached on a population level by individual Member States for the average population size occurring in its waters throughout the year.

Target setting

The proposed target or GES means that knowledge is required both on bycatch and on the population size, within appropriate confidence values. This poses problems, as has been demonstrated by ICES (2010). With the available data on bycatch of harbour porpoises it was not possible to conclude whether or not more than 1.7% of the population had been bycaught 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 (EU 812/2004).

Future steps necessary

There is clearly a lack of information on aspects of this indicator, although information is improving all the time. 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 35 and 36. Concerning bycatch, the following aspects should be further developed through linkages with appropriate fora:

1) Target (in %) for other species than the harbour porpoise.
2) Population (or management unit) against which to set the target (developed already for harbour porpoises in the North Sea), and development of a baseline for each population or management unit.
3) Development of a standardized monitoring methodology for bycatch or alternatively a mechanism for standardizing data post collection. This is currently being progressed through WGBYC.
4) Development of an assessment tool. This is currently being progressed through WGBYC.

Literature

ICES. 2011. Report of the Working Group for Bycatch of Protected Species (WGBYC), February 1–4 2011, Copenhagen Denmark. ICES CM 2011/ACOM:26. 75 pp.

ICES. 2010. ICES Advice on the EC request on cetacean bycatch Regulation 812/2004, Item 3. ICES Special Request Advice, Copenhagen, October 2010.

ICES. 2008. Report of the Working Group on Marine Mammal Ecology (WGMME), February 25–29 2008, St. Andrews, UK. ICES CM 2008/ACOM: 44. 83 pp.

Annex 5: Recommendations

Recommendation I

WGMME reiterates its **strong recommendation** that the bycatch management procedures developed under the SCANS-II and CODA projects (SCANS-II 2008; Winship, 2009; CODA, 2009) should be taken forward to develop management frameworks for bycatch at a European level. Without explicit conservation and management objectives, further development of bycatch management procedures is limited. It is proposed that WGMME and WGBYC collaborate to progress this approach as part of the MSFD indicator development during 2013.

Recommendation II

WGMME **strongly recommends** that Member States use the proposed management units for reporting requirements of the Habitats Directive and for the development of indicators and their assessment for the Marine Strategy Framework Directive. In summary, there is a single MU in European North Atlantic for common dolphin (*Delphinus delphis*), white beaked dolphin (*Lagenorhynchus albirostris*), white sided dolphin (*Lagenorhynchus acutus*) and minke whale (*Balaenoptera acutorostrata*). For bottlenose dolphin (*Tursiops truncatus*) there are ten separate units closely associated with the mainly resident inshore populations in the European North Atlantic and a separate MU for the wider ranging mainly offshore animals. For harbour porpoise (*Phocoena phocoena*), MUs are proposed for the Iberian Pennisula, Bay of Biscay, Celtic Sea (including SW Ireland, Irish Sea and Western Channel) and NW Ireland/West Scotland and the North Sea. The MUs for harbour porpoises may need to be revisited as indicators for MSFD become better defined and aligned with ICES rectangles to enable the calculation of more accurate bycatch estimates. For the purposes of MSFD, it maybe that consideration of the species will need occur at the regional seas level (e.g. North Sea).

Recommendation III

WGMME strongly supports the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2015 and **recommends** that it is supported by all range states and by ICES, ASCOBANS and the European Commission. Continuation of these surveys is essential for accurate population estimates, essential for reporting requirements of both the Habitats Directive and the Marine Strategy Framework Directive.

Recommendation IV

Following increasing understanding of harbour seal populations, WGMME reiterates its **recommendation** to OSPAR that the seal EcoQO subunits be updated. The revised harbour seal EcoQO should therefore read:

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 twelve sub-units of the North Sea. These sub-units are: Shetland; Orkney and north coast of Scotland; Moray Firth and East coast of Scotland; the Greater Wash/Scroby Sands; the French North Sea and Channel coasts; the Netherlands Delta area; the Wadden Sea; Heligoland; Limfjord; the Kattegat; the Skagerrak; the Oslofjord; and the west coast of Norway south of 62°N.

and for grey seals:

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. These subunits are: Orkney; Firth of Forth; the Farne Islands; the Greater Wash; the French North Sea and Channel coasts; the Wadden Sea; Heligoland; Kjørholmane (Rogaland).

Such a change in the subunits would more accurately reflect current monitoring and/or management areas. For the development of the MSFD indicators is recommended, however, that the subunits do not get specifically listed. Thus, avoiding the need to rewrite/update the wording of the indicator as new information on populations comes to light.

Recommendation V

WGMME **recommends** that power analyses are undertaken for all seal management areas to determine the trends in populations that can accurately be assessed with current monitoring practices. It is also recommended that ICG-COBAM give consideration to using the EcoQO approach rather than the more generic baseline proposed but that it would be useful to relate the percentage change to some earlier baseline such as the favourable reference population determined under the Habitats Directive.

Recommendation VI

It is **recommended** that WGMME assess the JCP outputs when they become available with a view to their contribution to international reporting requirements in 2013. The current Article 17 guidance for the 2013 reporting round includes a much greater emphasis on transboundary reporting where appropriate (European Commission, 2011). Further development/refinement of MSFD indicators of biodiversity will be required through 2013 and implementation of the monitoring needs to meet these requirements is needed by 2014. Development of an international equivalent of the JCP is also recommended that could be held by ICES.

Recommendation VII

The marine renewable industry is developing rapidly and regulators need to make decisions on granting consent for licensing in the near future. As the industry expands from a few sites to a large number of sites over larger areas of sea, it will become increasingly important to be able to predict population effects in order to meet management objectives such as Favourable Conservation Status under the Habitats Directive and Good Environmental Status under the Marine Strategy Framework Directive. A good management framework requires a sufficient level of basic understanding of animal-device interactions including a deploy and monitor strategy for assessing these interactions; it would also benefit from ongoing data collection (monitoring) at appropriate scales to allow the incorporation of a feedback mechanism and to enable determination of whether management actions are allowing objectives to be met. *The WGMME recommends the development of an appropriate precautionary management framework for marine renewable energy technologies.*