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

9–12 February 2015

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Executive summary

New information on distribution and abundance of harbour porpoise available from aerial surveys in the North Sea has been compiled and will be used in project DE-PONS (Disturbance Effects on the Harbour Porpoise Population in the North Sea) to identify areas with high porpoise densities and to predict seasonal distribution and density. New information on abundance and trends available for coastal bottlenose dolphins in Scottish and Welsh waters in the UK, in waters west of Ireland, off the Normano-Breton coast of France, and off the north coast of Spain has been collated together with new information on sperm whales and short-finned pilot whales in the Canary Islands. Updated or new information on distribution and abundance of several cetacean species was available from extensive coastal and offshore surveys off France, mainland Portugal and Madeira. Large-scale cetacean surveys are planned for the central and eastern North Atlantic in summer 2015 (NASS), and in European Atlantic waters in summer 2016 (SCANS-III). Plans for a Mediterranean cetacean survey continue to be pursued.

New results on population structure, available for harbour porpoise and bottlenose dolphin have been compiled. Satellite telemetry and static acoustic monitoring data were used to assign boundaries between populations in the North Sea-Skagerrak, the Belt Sea and the Baltic proper. New results from a genetic analysis of harbour porpoise tissue from Iberia, northern Europe and Turkey indicate a level of differentiation of the Iberian population that may warrant categorisation as a separate subspecies. New results from genetic, stable isotope and diet studies indicate that bottlenose dolphin population structure is hierarchical in the Northeast Atlantic, comprising coastal and pelagic ecotypes. The coastal ecotype comprises a north and south population and there are pelagic ecotypes in the Atlantic and Mediterranean. No new information was available on management frameworks.

A threat matrix was completed for the main marine mammal species in each regional seas area. While fishery bycatch is a significant concern, especially for harbour porpoise, common dolphin, coastal bottlenose dolphin and ringed seal, contaminants are also a major concern, especially for harbour porpoise, killer whale and bottlenose dolphin. In the Baltic Sea, contaminants and habitat degradation are a serious concern for all resident marine mammal species. In the Bay of Biscay/Iberian Peninsula and Macaronesia, sonar is a significant threat to beaked whales, and in the former area fin whale and sperm whale are threatened by collisions with shipping. The small population of Mediterranean monk seal in Madeira is threatened by habitat degradation, disturbance and deliberate killing. Text on marine mammals has been provided for the ICES Ecosystem Overviews.

Where their distributions overlap, there is some evidence of negative ecological interactions between harbour seals (*Phoca vitulina*) and grey seals (*Halichoerus grypus*). There is spatial variation in their populations trajectories: in some regions both species are increasing (e.g. Wadden Sea, Baltic Sea and Kattegat, Ireland, France) while in other regions harbour seals are declining while grey seal numbers are on the rise (e.g. North Sea, Orkney, Sable Island). Potential interactions (at-sea distribution, competition for prey, haul-out site use, and predation of harbour seals by grey seals) were reviewed. Recent evidence of direct predation of harbour seals by grey seals in the North Sea was highlighted, as well as evidence of predation of harbour porpoises by grey seals in the same region.

The ICES seals database was updated with limited data from few countries. It is anticipated that the database will be fully updated in 2015 to contribute to OSPAR's Intermediate Assessment in 2017.

Marine mammals have been included in whole ecosystem models (e.g. EwE) and in minimum realistic models (e.g. GADGET), among others, in studies principally focused on trophic relationships, resource competition between fisheries and marine mammals, and consequences for fish stocks. There is the potential to add fishery by-catch mortality of marine mammals to such models although few examples exist where this has been done. Other types of biological interaction (e.g. parasite transmission) have been less well covered. All models have limitations and some kind of validation exercise is essential to confer credibility on the predictions.

1 Introduction

The Working Group on Marine Mammal Ecology (WGMME) met at the Institute of Zoology, Zoological Society of London (ZSL), Regent's Park, London, from 09 to 12 February 2015. The list of participants and contact details are given in Annex 1.

On behalf of the working group, the chairs would like to thank the efforts and support provided by both Paul Jepson (ZSL) and Rob Deaville (ZSL). In particular, their help in organising the meeting, organising the free entry into London Zoo and organising a porpoise necropsy we could attend. WGMME participants could not have wished for more!

The Chairs also acknowledge the diligence and hard work of all the participants before, during and after the meeting, which ensured that the Terms of Reference were addressed.

2 Terms of Reference 2015

The meeting Agenda appears in Annex 2. The following Terms of Reference (ToRs) and the work schedule were adopted on 9 February 2015:

- a) Review and report on any new information on population abundance, population/stock structure and management frameworks for marine mammals;
- b) Compile a matrix of threats to the predominant cetacean species in each of the MSFD regional seas. Consider ways in which this information could be incorporated into the ICES advice;
- c) Review North Atlantic information on negative and positive ecological interactions between grey seal (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) populations;
- d) Update the database for seals, and review its usefulness in meeting the needs of Member States under the EU's MSFD;
- e) Review and evaluate multispecies models that incorporate marine mammal consumption to assess marine mammal impacts on fishery resources, and make recommendations for improvements in input data and assumptions for the North Atlantic.
- f) Produce four short paragraphs for the ICES Ecosystem Overviews on the state of sea mammal populations, one paragraph for each of the following ICES ecoregions: Greater North Sea, Celtic Seas, Bay of Biscay and the Iberian coast and Baltic Sea.

WGMME will report to the attention of the Advisory Committee (ACOM) by 4 April 2015.

3 ToR a: Review and report on any new information on population abundance, population/stock structure and management frameworks for marine mammals

New information on cetacean abundance, including distribution, and population structure is described below. Information on seals is included under ToR (c). No new information on management frameworks was available.

3.1 Abundance and trends of coastal cetaceans in the UK

Bottlenose dolphins in Scottish waters

Bottlenose dolphins have been studied using photo-identification along the east coast of Scotland since 1989. In recent years, a number of analyses have updated knowledge of abundance and trends in this area. The total east coast population was estimated to be 195 animals (95% highest posterior density interval (HPDI): 162–253) in 2006–2007 (Cheney *et al.*, 2013). Analysis of trends in abundance has shown that the number of animals using the Moray Firth Special Area of Conservation (SAC), NE Scotland, in summer has remained more or less stable but that the total east coast population is likely to be increasing (Figure 3.1; Cheney *et al.*, 2014). The number of animals present in summer in the Tayside and Fife area of SE Scotland (the southern end of the range of the population) has been estimated to be between 71 (95% CI: 63–81) and 91 (95% CI: 82–100) individuals in the period 2009–2013 (Arso Civil, 2014; Quick *et al.*, 2014).

Off the west coast of Scotland, the bottlenose dolphin population was estimated to be 45 animals (95% HPDI: 33–66) in 2006–2007 (Cheney *et al.*, 2013).

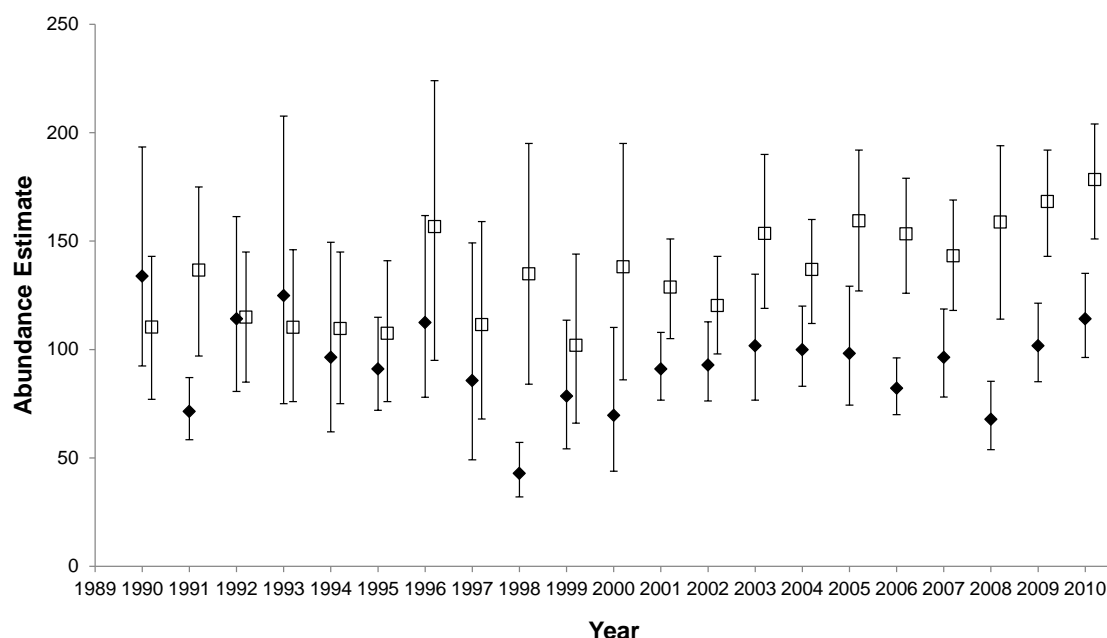


Figure 3.1. Annual estimates of the number of bottlenose dolphins using the Moray Firth Special Area of Conservation from 1990 to 2010 (black diamonds) with 95% confidence intervals and of the total east coast of Scotland bottlenose dolphin population with posterior means (clear squares) and 95% highest posterior density intervals. From Cheney *et al.* (2014).

Bottlenose dolphin and harbour porpoise in Welsh waters

In Wales, line transect surveys for cetaceans and photo-identification surveys of bottlenose dolphins continue to be conducted in the Cardigan Bay SAC and the wider Cardigan Bay. Feingold and Evans (2014) present new estimates of abundance for 2011–2013 and updated information on longer term trends.

From line transect surveys, bottlenose dolphin abundance in Cardigan Bay SAC was estimated at 133 individuals (CV = 29.5%) in 2011, 70 (CV = 33.0%) in 2012, and 90 (CV = 35.6%) in 2013, concentrated in the coastal zone. Abundance in the entire Cardigan Bay was estimated to be 309 (CV = 28.3%) in 2011, 390 (CV = 24.9%) in 2012, and 254 (CV = 26.8%) in 2013. Harbour porpoise abundance in Cardigan Bay SAC was estimated at 340 individuals (CV = 46.4%) in 2011, 169 (CV = 29.1%) in 2012, and 146 (CV = 21.3%) in 2013. Abundance estimates for porpoises in the entire Cardigan Bay were much higher: 1,074 (CV = 28.7%) in 2011, 565 (CV = 20.4%) in 2012, and 410 (CV = 20.4%) in 2013.

Robust design mark–recapture estimates of the number of bottlenose dolphins using Cardigan Bay SAC between 2001 and 2013 have ranged from 77 (in 2002) to 168 (in 2012). Examination of the trend shows an initial increase to 2007 but a decline towards the end of the time-series. Estimates for the last three years (2011–2013) were 147, 168 and 101, respectively.

Mark–recapture estimates of the number of bottlenose dolphins using the entire Cardigan Bay are available from 2005 and have ranged from 128 (2005) to 232 (2012). Examination of the trend indicated an initial increase, with numbers peaking around 2009, and a subsequent decline. Estimates for the last three years (2011–2013) were 193, 232 and 167, respectively.

3.2 Common and bottlenose dolphin abundance west of Ireland

Inshore boat surveys off northwest, west and southwest Ireland were carried out between August and October 2010 generating a total of 92 sightings of 528 individuals of at least five species (Ryan *et al.*, 2010). Density and abundance estimates were derived for common dolphins in the west survey area (density = 4.56 individuals/km²; abundance = 5254; CV = 0.44) and in the southwest survey area (density = 2.44 individuals/km²; abundance = 2812; CV = 0.45).

A survey of bottlenose dolphins in the Lower River Shannon candidate SAC (cSAC) undertaken in 2010 resulted in an estimate of 107 (CV=0.12) individuals (Berrow *et al.*, 2010). Previous estimates ranged from 114 in 2008 to 140 in 2006, suggesting that the population of bottlenose dolphins in the Lower River Shannon cSAC is relatively stable.

3.3 Harbour porpoise distribution and abundance in German waters

Under the framework of the Natura 2000 monitoring programme in Germany, four aerial line-transect surveys for marine mammals were conducted between March and July 2014 (Viquerat *et al.*, 2015).

In March/April and May 2014 the Borkum Reef Ground (areas DE and DW; Figure 3.2), southwestern German Bight, was surveyed; over 2790 km of transect a total of 320 harbour porpoise sightings of 349 individuals (no calves) were recorded. The distribution of estimated density in areas DE and DW is shown in Figure 3.3. Estimated density was significantly lower in March/April, with 0.52 animals/km²

(CV=0.29), than in May 2014 (1.33 animals /km²; CV=0.28). The data from the southern German Bight reflect the general increase in porpoise density in the overall southern North Sea.

In June and July 2014 the Sylt Outer Reef (area CN; Figure 3.2) in the northeastern German Bight was surveyed; over 3024 km of transect a total of 288 harbour porpoise sightings of 370 individuals (including 31 calves) were recorded. The distribution of estimated density in area CN is shown in Figure 3.4. Estimated density was lower in June 2014 (0.95 animals /km², CV=0.33) than in July 2014 (1.36 animals /km², CV=0.28) and 2014 estimates were lower compared to estimates from other years. Nevertheless, no significant trend has been found.

A long-term analysis of the presence of mother-calf pairs in the southern German Bight has shown a significant increase in calves (Viquerat *et al.*, 2015). Comparison of the spatial distribution of calves in 2002–2007 and in 2008–2012 shows a rise in use of the Borkum Reef Ground by mother-calf pairs since 2008. This reinforces the importance of the region as a potential breeding habitat in addition to the main concentration of mother-calf pairs in Sylt Outer Reef.

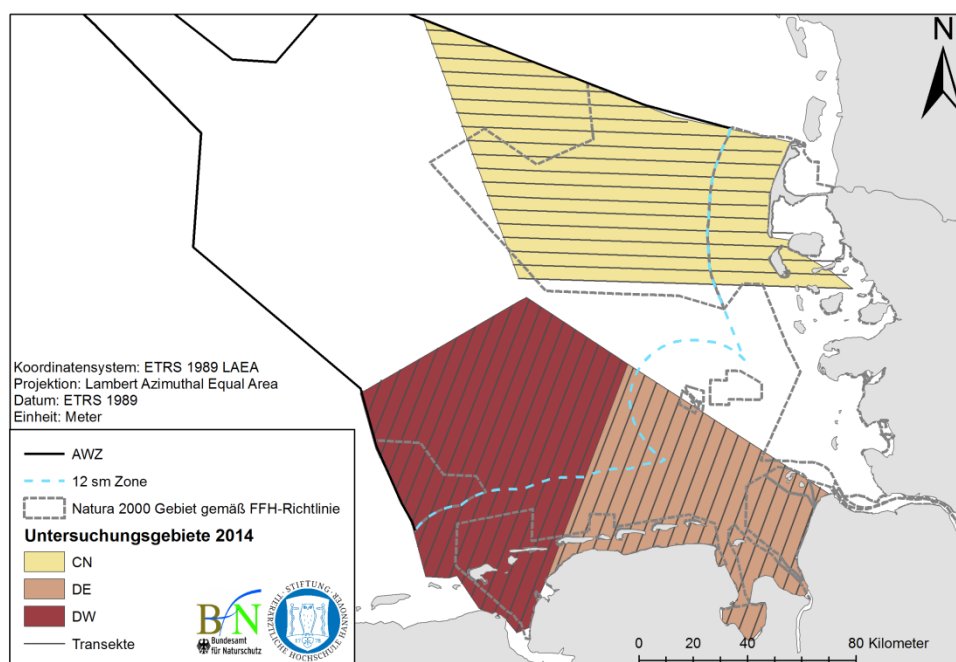


Figure 3.2. Survey blocks in the German North Sea in 2014. Parallel transects are spaced 5 km apart (from Viquerat *et al.*, 2015).

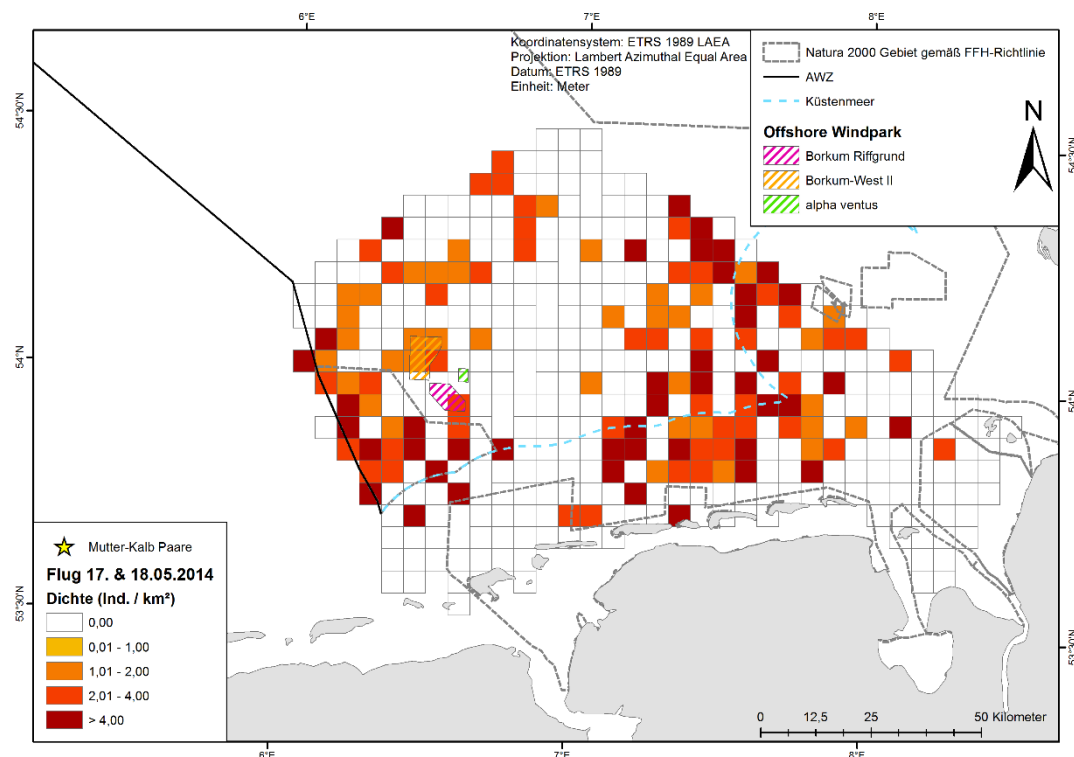


Figure 3.3. Spatial distribution of harbour porpoise density (animals/km²) during the aerial survey in May 2014 in areas DE and DW. Grid cell size: 5x5 km. From Viquerat *et al.* (2015).

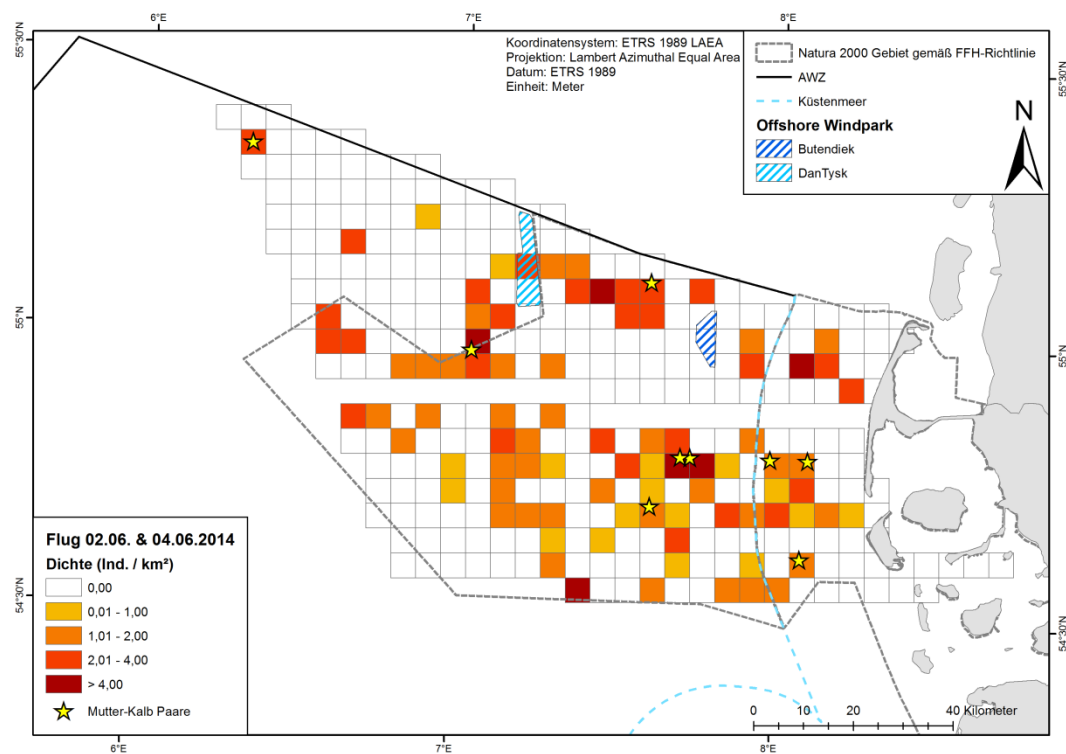


Figure 3.4. Spatial distribution of harbour porpoise density (animals/km²) during the aerial survey in June 2014 in area CN. Grid cell size: 5x5 km. From Viquerat *et al.* (2015).

3.4 Harbour porpoise abundance in Dutch waters

Aerial surveys to estimate the abundance of harbour porpoise on the entire Dutch continental shelf were conducted during 11–16 July 2014. Surveys were conducted along predetermined track lines using distance sampling methods corrected to account for animals missed on the transect line. The Dutch continental shelf was stratified into four areas: A - Dogger Bank, B - Offshore, C - Frisian Front and D - Delta (Figure 3.5).

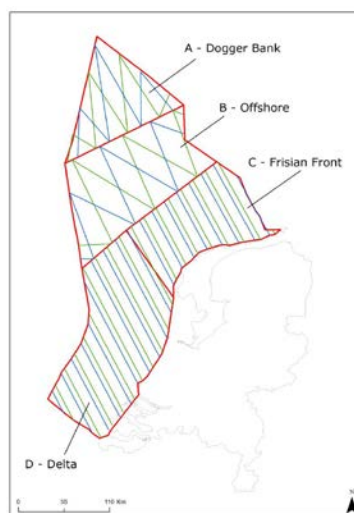


Figure 3.5. Map of the Dutch continental shelf with the planned track lines in areas A - Dogger Bank, B - Offshore, C - Frisian Front and D - Delta. Colours indicate sets of track lines.

In total, 229 sightings of 273 individual harbour porpoises were made (Figure 3.6). Estimated porpoise densities varied between 0.37–3.08 animals/km² in areas A–D (Table 3.1). The average density over the entire Dutch continental shelf was 1.29 animals/km². The total abundance of harbour porpoises on the Dutch continental shelf (areas A–D) in July 2014 was estimated to be ~77 000 animals (95% CI: 43 000–154 000). The total estimate for July 2010 was lower at ~26 000 animals (95% CI: 14 000–54 000); however, the 95% confidence intervals of these two estimates overlap (Table 3.1).

In total, 28 sightings of other marine mammal species were made, mostly of unidentified seals. Three individual minke whales were seen in the Dogger Bank area on 12 July 2014. One group of three white-beaked dolphins was recorded in the same area.

Table 3.1. Comparison between density and abundance estimates of Harbour Porpoises obtained in July 2010 and July 2014 using results from Geelhoed *et al.* (2011; 2013a).

	DENSITY (ANIMALS/KM ²)	95% CI	ABUNDANCE (ANIMALS)	95% CI	CV
2014					
A	3.08	1.50–6.45	29 689	14 375–61 995	0.37
B	0.37	0–1.21	6297	0–20 509	0.96
C	1.83	0.97–4.11	22 010	11 623–49 439	0.39
D	0.90	9.46–1.84	18 778	9548–38 167	0.36
Total	1.29	0.73–2.60	76 773	43 414–154 265	0.34
2010					
A	0.40	0.18–0.85	3806	1738–8165	0.40
B	0.48	0.21–1.06	8055	3589–17 872	0.42
C	0.34	0.05–0.89	4039	553–10 701	0.62
D	0.48	0.21–1.06	10 098	4341–22 024	0.40
Total	0.44	0.24–0.90	25 998	13 988–53 623	0.34

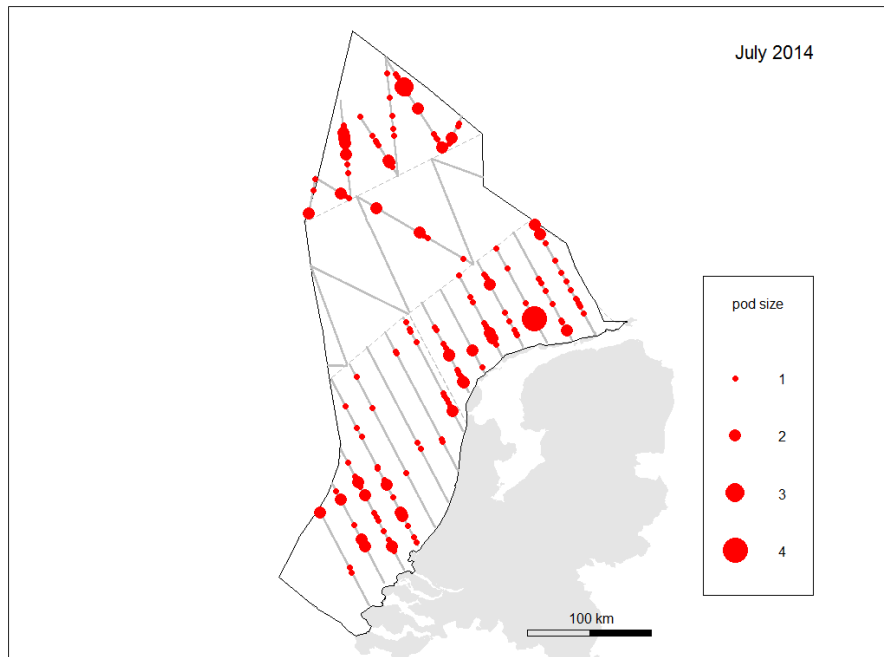


Figure 3.6. Realised survey effort and harbour porpoise sightings during the Dutch continental shelf survey in July 2014.

3.5 Harbour porpoise in Belgian waters

Aerial line transect surveys continue to be conducted by the Royal Belgian Institute of Natural Sciences (RBINS). Estimates of the average density of harbour porpoises in an area covering most Belgian waters in 2013 were: 0.84 animals/km² in January, 0.94 animals/km² in February, 1.72 animals/km² in May and 0.61 animals/km² in September. These estimates have wide confidence intervals.

In April 2014 a highest-ever average density over this survey area of 3.96 (3.21–4.88) animals/km² was estimated (data RBINS, unpublished).

Tens of porpoises were observed in inland waters (Scheldt and tributaries) during late winter/spring 2013 (peaking between 27 March and 20 June); an occurrence that has not been observed in recent decades (www.waarnemingen.be; Haelters, 2013). This remarkable incursion far upstream may have been related to a relatively long and cold winter, with a possible prey shortage in coastal waters, combined with an increasing influx of anadromous fish such as smelt, *Osmerus eperlanus* (Haelters, 2013).

3.6 Project DEPONS

A subproject within the DEPONS project (Disturbance Effects on the Harbour Porpoise Population in the North Sea; www.depons.au.dk) aims to identify areas in the eastern, central and southern North Sea with high porpoise densities and to predict the seasonal distribution and density of porpoises. Data from the large-scale Small Cetaceans in the European Atlantic and North Sea (*SCANS-II*) survey (Hammond *et al.*, 2013) are being combined with smaller scale but higher temporal resolution datasets from dedicated and standardised aerial line transect surveys from Denmark, Germany, the Netherlands and Belgium (Gilles *et al.*, 2009; 2011; 2012; Haelters *et al.*, 2011; 2013; Scheidat *et al.*, 2012; Geelhoed *et al.*, 2013; 2014).

More than 167 000 km of survey effort data with more than 14 800 sightings of harbour porpoise, collected between 2005 and 2013, are being analysed together using spatial statistical models for the first time. Results will be available for the southeastern North Sea in spring and autumn/winter and for the whole North Sea in summer. An example of preliminary results of the modelling is given in Figure 3.7. The model will be further refined in 2015 and final results are expected to deviate slightly from those in Figure 3.7. This project aims to demonstrate the value of combining coordinated small-scale surveys using a standardised methodology with large-scale surveys to inform trends in abundance and distribution patterns.

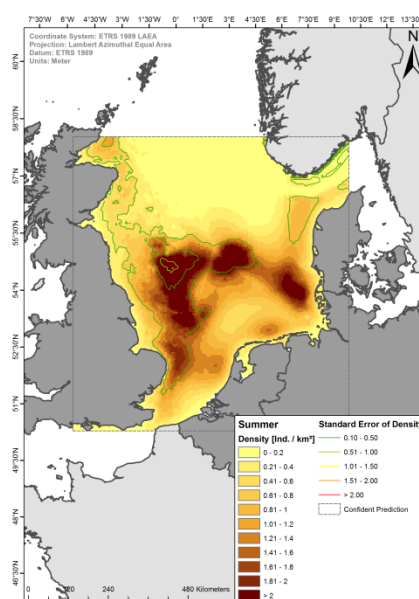


Figure 3.7. Example of preliminary results of modelled densities for harbour porpoises in the North Sea in summer 2005–2013.

3.7 Cetacean distribution and abundance in French waters

Updated results from project SAMM

WGMME (2014) reported on dedicated aerial surveys across all waters under French jurisdiction and some adjacent areas: *Suivi Aérien de la Mégafaune Marine (SAMM)*. These surveys, conducted in winter 2011–2012 and summer 2012, were designed to document distribution of all megafauna (mammals, birds, turtles, and large fish) visible from the air and assess relative abundance to look at distributional changes in time and space. Total survey effort was 48 600 km in winter and 53 200 km in summer (see Figure 3.7 of WGMME 2014). The project was completed in November 2014 (Pettex *et al.*, 2014). The survey used a single platform and did not employ the circle-back methodology so estimates of abundance were not corrected for perception bias. However, they were corrected for availability bias using surfacing time data from the literature.

Estimates of harbour porpoise abundance were identical in the Channel in both seasons (~26 000 individuals) whereas in the Atlantic, the estimate in summer (~20 000 individuals) was four times greater than in winter (~4600 individuals; Table 3.2). Predicted densities showed an aggregation of harbour porpoise in the eastern Channel and southern North Sea in winter but low densities all along the coast down to the southern Bay of Biscay (Figure 3.8). In the summer, two areas of high predicted density were found: the southern North Sea and a broad region composed of the Celtic Sea, northern Bay of Biscay and western Channel.

Common and striped dolphins were considered jointly because there were too many sightings that could not be identified at species level with any certainty. Estimated abundance varied from 290 000 individuals in winter to 690 000 in summer (this difference not significant) (Table 3.2). A small fraction was predicted to be present in the western Channel during winter. Predicted habitat in winter included shelf waters of the Bay of Biscay, the western Channel and the Celtic Sea, predominantly parallel with the shelf break (Figure 3.9). In summer, highest densities were predicted over the slope and in oceanic waters.

No seasonal difference in bottlenose dolphin abundance was found (18 000 individuals in winter vs. 11 000 in summer) (Table 3.2). Predicted densities followed the slope in both summer and winter and were the highest over the slope of the southern Bay of Biscay in winter (Figure 3.10). Lower densities were predicted across shelf habitats in the Celtic Sea, western Channel and Bay of Biscay.

Long-finned pilot whales and Risso's dolphins were estimated in the low thousands in both seasons (Table 3.2), with a distinctive preference for slope habitats (Figure 3.11). Fin whales were estimated at ~2800 individuals in the summer in oceanic waters of the Bay of Biscay, where low hundreds were also found in winter (Table 3.2). Beaked whales, mostly *Ziphius cavirostris*, were estimated in both seasons in the thousands (Table 3.2).

Table 3.2. Estimated abundance (not absolute) of the main cetacean taxa in Atlantic (282 140 km²) and Channel (92 845 km²) waters.

Taxon	Season	ATLANTIC			CHANNEL		
		Estimated abundance	95% Confidence Interval		Estimated abundance	95% Confidence Interval	
Harbour porpoise	Winter	4643	2466	8742	26 556	16 797	41 984
	Summer	19 928	12 639	31 819	26 417	18 946	36 833
Common and striped dolphins	Winter	286 413	185 280	452 233	13 484	10 763	17 493
	Summer	694 805	493 104	986 598	1209	398	3671
Bottlenose dolphin	Winter	17 694	8356	37 524	1412	530	3760
	Summer	10 937	5907	20 834	2317	896	5992
Fin whale	Winter	372	86	1660	76	15	394
	Summer	2786	1396	5607	291	98	863
Risso's dolphin	Winter	1373	310	6274	229	55	947
	Summer	2461	851	7283	84	15	467
Long finned pilot whale	Winter	2886	1276	6951			
	Summer	3864	2064	7611			
Beaked whales	Winter	3269	1297	8238			
	Summer	7002	3055	17 318			

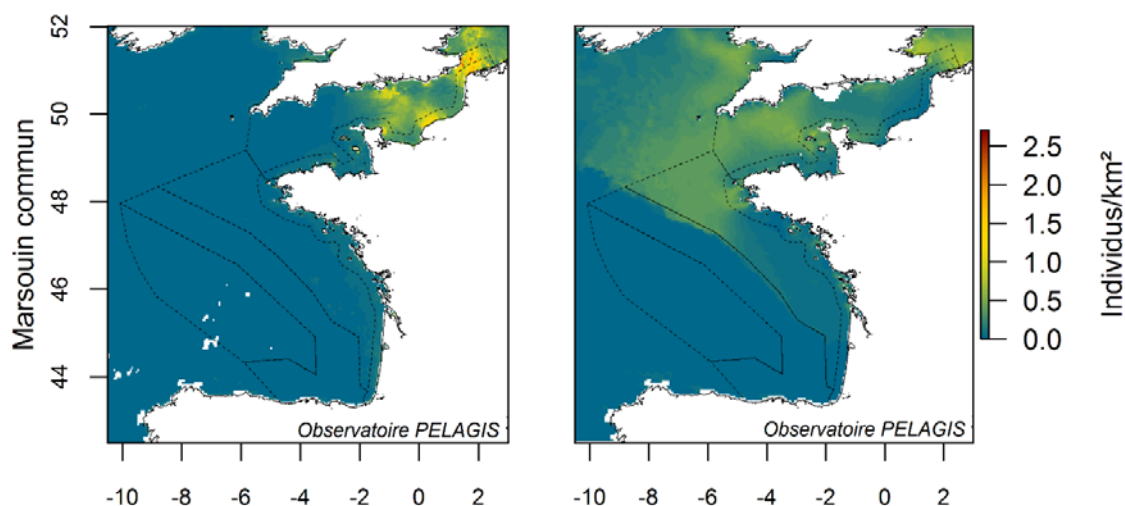


Figure 3.8. Predicted habitat use of harbour porpoise in winter (left) and summer (right). The lines delineate different survey strata in the Bay of Biscay.

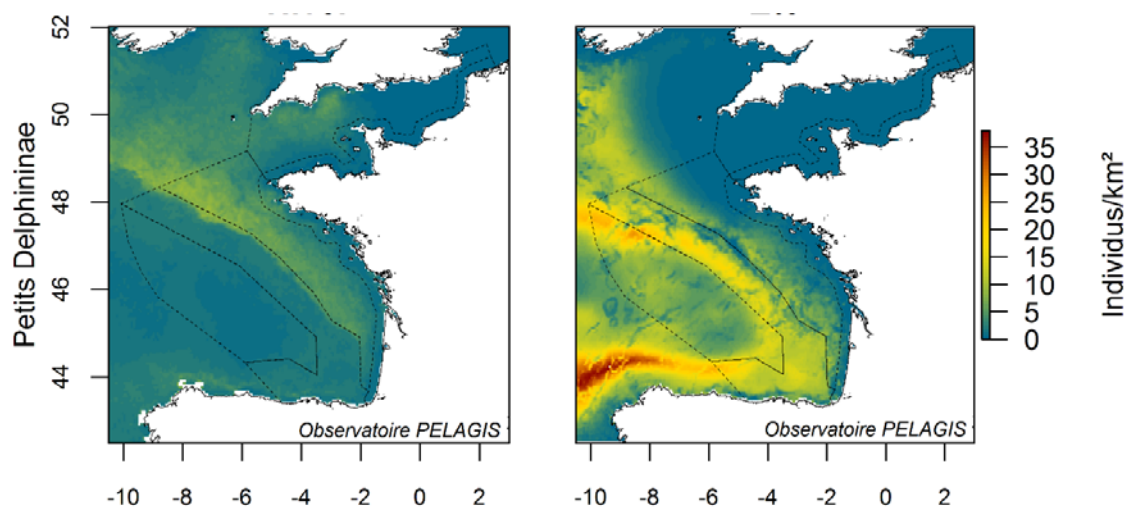


Figure 3.9. Predicted habitat use of common/striped dolphins in winter (left) and summer (right).

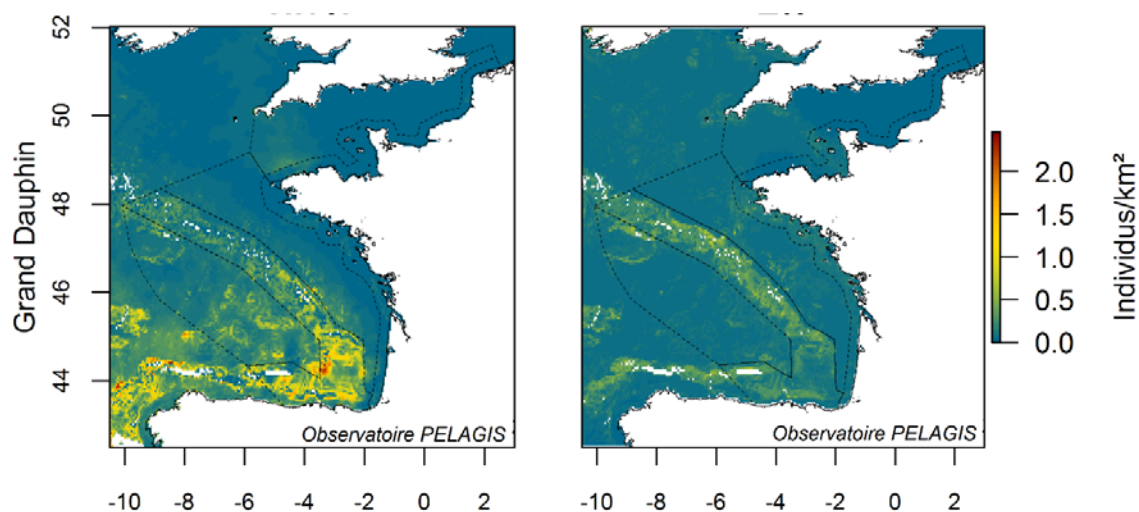


Figure 3.10. Predicted habitat use of bottlenose dolphin in winter (left) and summer (right).

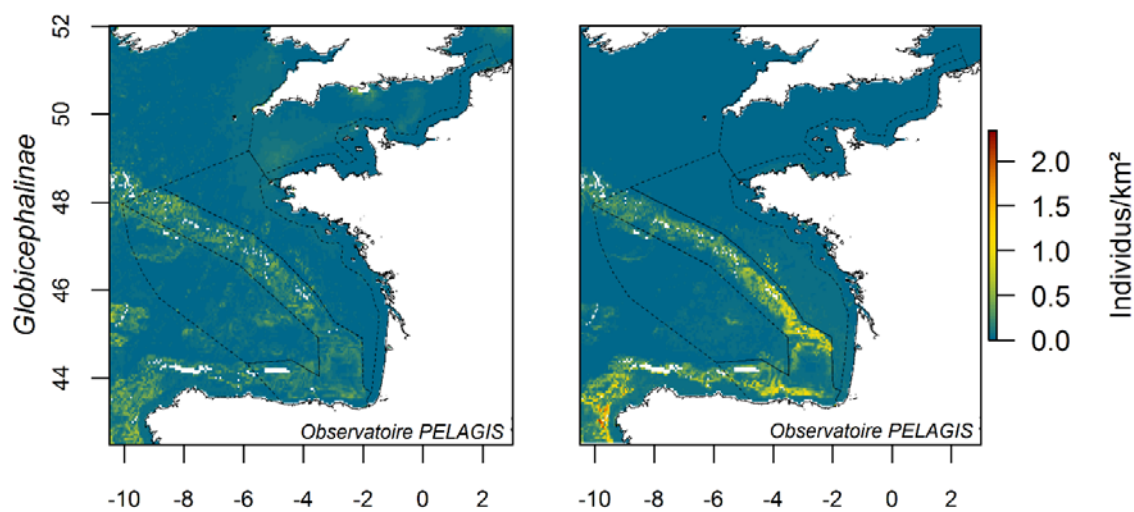


Figure 3.11. Predicted habitat use of long-finned pilot whales in winter (left) and summer (right).

Bottlenose dolphins in the Normano-Breton gulf

The Normano-Breton Gulf population of bottlenose dolphins was estimated in summer 2010 as 420 animals (95% CI: 331–521) using mark-recapture analyses on photo-identification data. Biopsy samples were collected from 92 individuals between 2010 and 2012 and indicated that the dolphins are part of one genetic population. Stable isotopes analyses indicated that they are organized in three ecological clusters that are consistent with social clusters (Louis, 2014).

3.8 Cetaceans in Spanish waters

Using photo-identification data obtained over an eight-year period (2003–2010) abundance of bottlenose dolphin and long-finned pilot whale in the Basque country has been estimated to be 1579 individuals (95% CI 918–2830) and 1737 (95% CI 1236–2514), respectively (Marcos-Ipiña and Salazar, 2013).

Saavedra *et al.* (2014) developed population models for common dolphins, using mortality-at-age curves derived from stranding data obtained in Galicia (NW Spain) using Siler and Heligman-Pollard models to estimate an annual rate of decline of 8.8%, likely due to an unsustainably high bycatch rate.

Goetz *et al.* (2014) used a variety of data sources to assesses cetacean distribution, habitat preferences and determine potential hot spots for cetacean–fishery interactions in Galicia (NW Spain) and Portugal. The results suggest that cetacean occurrence is linked to prey distribution and that interactions with fisheries are most likely for common dolphins (coastal purse-seines and offshore trawls), bottlenose dolphins and harbour porpoises (coastal nets).

Molecular genetic studies have previously revealed a clear separation of the Iberian harbour porpoise from animals in more northerly Northeast Atlantic waters (Fontaine *et al.*, 2007; 2010; 2014), although there is also evidence that the two populations may be admixing in the Bay of Biscay (Alfonsi *et al.*, 2014). Recent results from microsatellite DNA analysis of harbour porpoise tissue samples from Iberia (n=189), northern Europe (n=113), and Turkey (n=16) and their combination with previously published sequences indicate a level of differentiation of the Iberian population that may warrant categorisation as a separate subspecies (Llavona *et al.*, 2014).

Sperm whales in the Canary Islands

An acoustic line-transect survey was conducted in offshore waters around the Canary Islands in autumn-winter 2009/2010 over an area of 52 933 km² to estimate the abundance of sperm whales (Figure 3.12; Fais, 2015). The absolute abundance of sperm whales was estimated to be 220 sperm whales (117–413, CV = 0.32). This estimate incorporated an estimate of $g(0) = 0.92$, calculated from data on sperm whale acoustic and diving behaviour obtained from DTAGs.

The lack of data on-site fidelity and connectivity of sperm whale groups inhabiting the Canary Islands precludes an accurate evaluation of the risk posed by ship-strikes for this species in the archipelago. Many areas of higher whale density within the archipelago were found to be consistent with those previously described, suggesting that these are important habitats for females and immature whales.

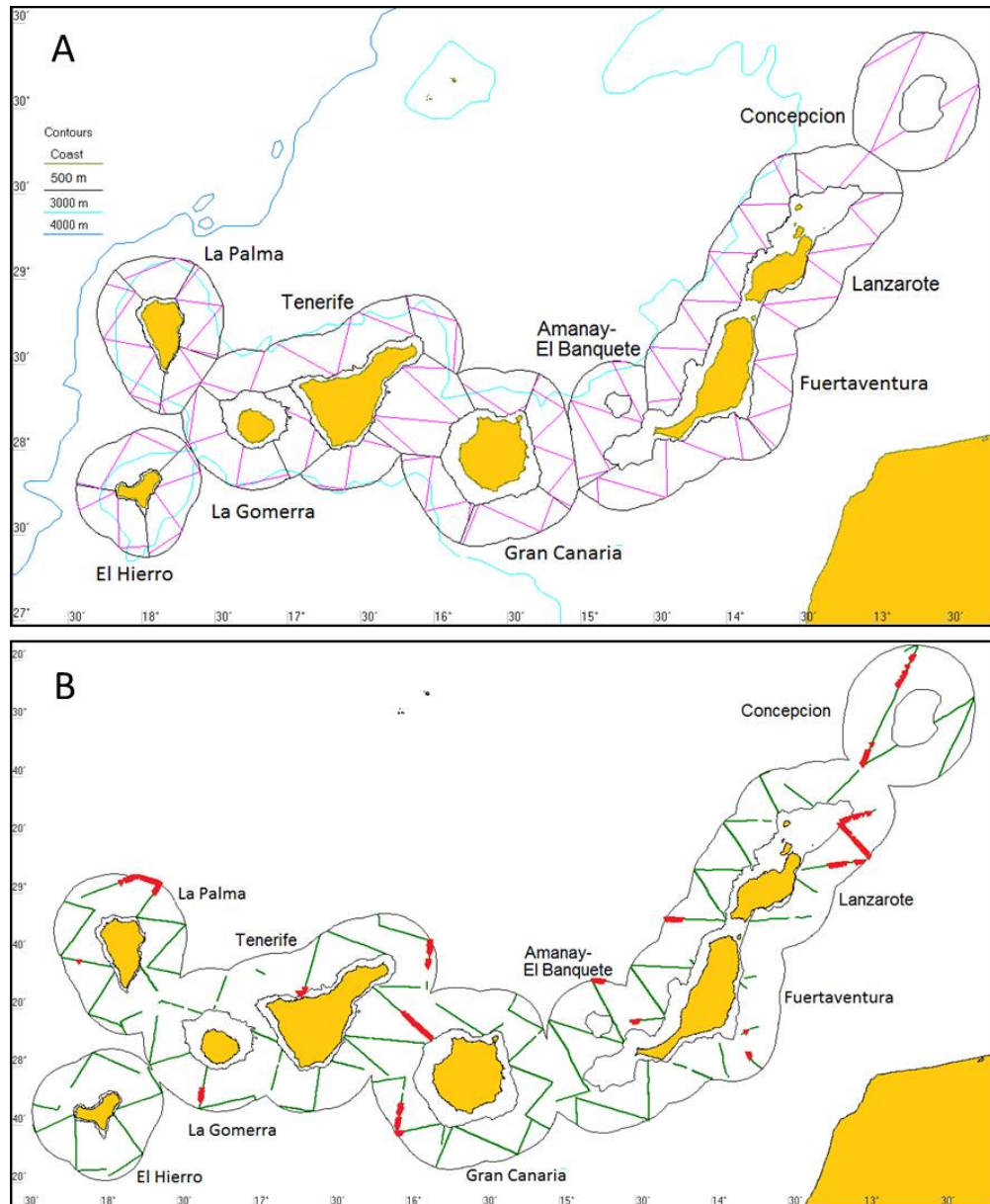


Figure 3.12. A) Survey area in the Canary archipelago indicating the names of the islands and seamounts, the survey blocks (black lines) and the designed transects (magenta lines). B) Acoustically surveyed tracks (green lines) and sections of transects from which sperm whales were acoustically detected (red triangles).

Distribution and abundance of short-finned pilot whales in the Canary Islands

Between 1999 and 2012, the Society for Cetacean Study in the Canary Archipelago (SECAC) carried out 1782 days of survey in the Canary Archipelago, focusing mainly on the islands of Tenerife, Gran Canaria, La Gomera and, since 2005, Lanzarote and Fuerteventura. A total of 59 231 nm were covered of which 38 131 nm were spent on survey effort (Figure 3.13). During the surveys, a total of 4242 sightings of 23 identified cetacean species were recorded. Short-finned pilot whale was the most frequently seen species, with 1081 sightings distributed as shown in Figure 3.14. In total, 757 pilot whale sightings were recorded on effort.

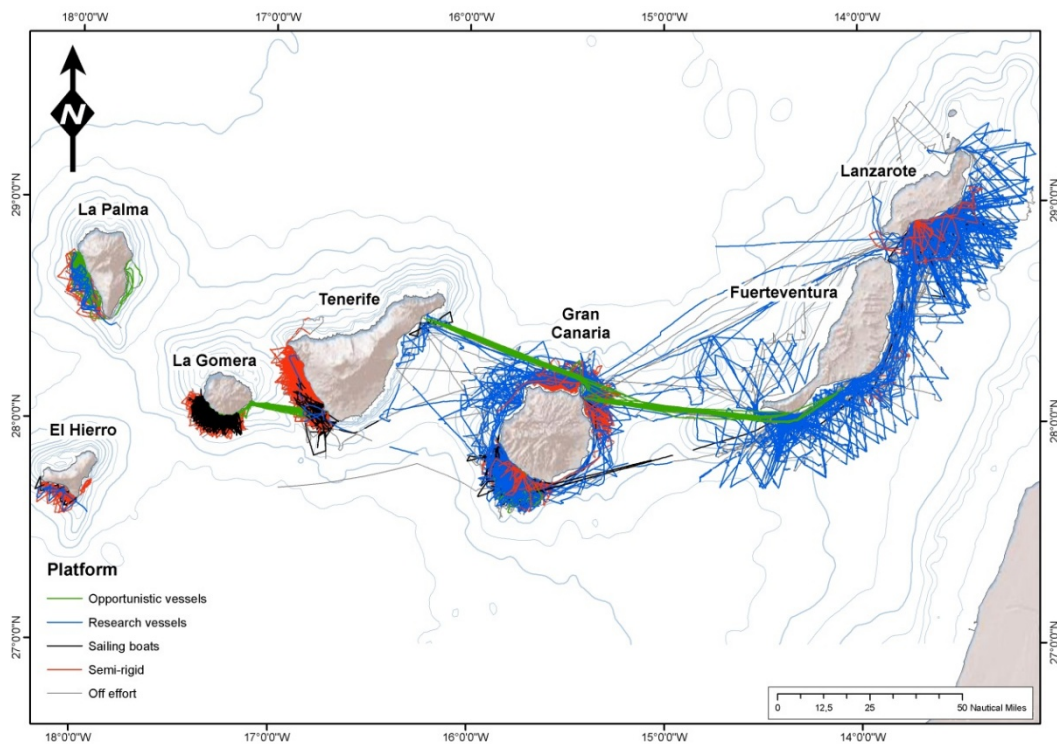


Figure 3.13. Track-lines covered on effort since 1999 around the Canary Islands on board various vessel types: semi-rigid inflatable boats (red); dedicated motor boats (blue); sailing boats (black); and opportunist vessels such as whale-watching boats and ferries (green). Off-effort track-lines are indicated in grey.

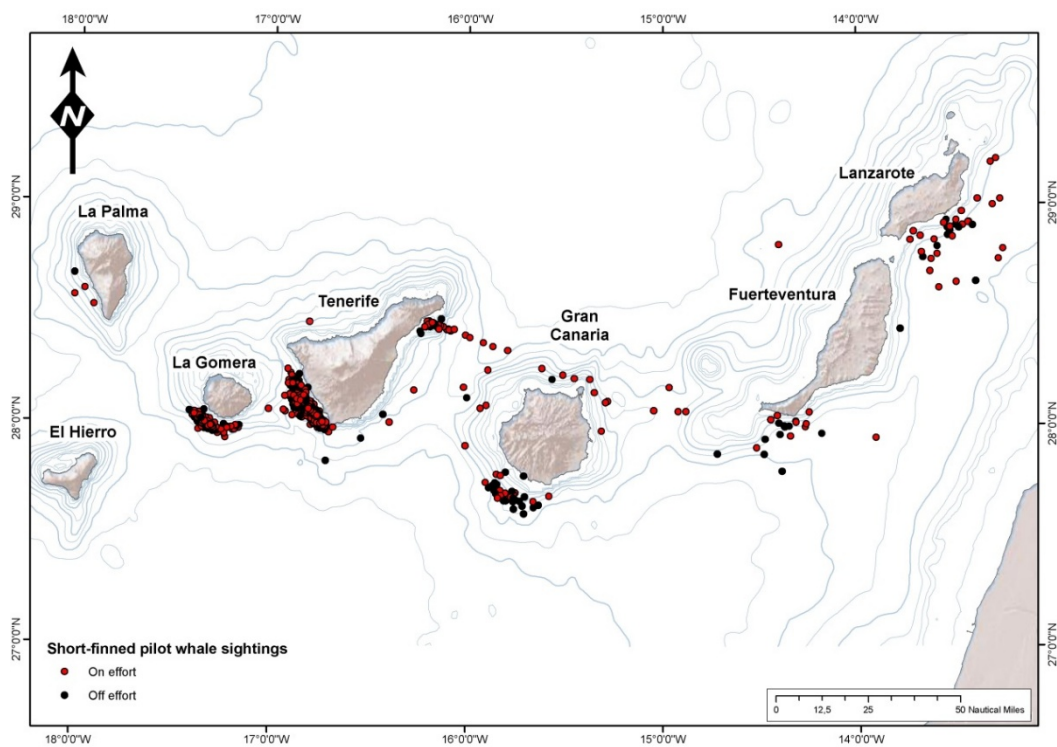


Figure 3.14. Short-finned pilot whale sightings on and off effort, recorded between 1999 and 2012 in the Canary Islands.

Short-finned pilot whale distribution and abundance was modelled using Generalised Additive Models (GAMs). Areas of high usage were predicted in areas downwind of islands at depths of 800–1500 m, deeper waters around 1500 m during summer and shallower depths closer to the coast (1000 m) during winter. The overall population in the Canary archipelago was estimated to be 1980 individuals (CV=0.32, 95%CI=1441–2324). Around Tenerife, abundance was estimated to be 1223 whales (CV=0.31; 95%CI=1046–1267) (Servidio, 2014).

The surveys included photo-identification of short-finned pilot whales resulting in the creation of an extensive catalogue for this species in the Canary Islands of 3275 identified individuals, of which 1310 were well-marked whales. Mark-recapture analysis of data from 2007–2009 estimated 636 (CV=0.028; 95%CI=602–671) resident animals and a total of 2698 individuals (CV=0.14; 95%CI=2057–3538) using the area southwest of Tenerife.

3.9 Cetacean abundance in Portuguese waters

Aerial surveys have been conducted under the Life MarPro project in 2010–2014 out to 50 nm from the Portuguese coast, covering 3322 km of survey effort over an area of 62 716 km². In 2013, 185 sightings of 2557 observed individuals of 13 species were recorded. In 2014, twelve species were recorded, with 181 sightings of 4732 individuals. Estimates of abundance calculated from these data are given in Table 3.3 (Santos *et al.*, 2012 revised in 2014 with new data from 2013). Estimates from ship surveys conducted under the same project are given in Table 3.4 (Santos *et al.*, 2012).

Between 2012 and 2013, 49 surveys with a total searching effort of 132 hours were made along the Portuguese coast (Sesimbra, Sado Estuary, Lisbon and Peniche), with 64 sightings of cetaceans, (Brito *et al.*, 2014). Most sightings of bottlenose dolphin were of individuals of the resident Sado Estuary population where the sampling effort was focused (Martinho *et al.*, 2014).

Table 3.3. Total abundance and relative density in 2010-2013 estimated from aerial surveys up to 50 nm from the coast for species for which it was possible to estimate abundance, with correction for availability bias. NE – Not estimated.

SPECIES	ABUNDANCE	% CV	95% CI	DENSITY
2010				
Minke whale	360	72%	99–1310	0.00574
Fin whale	90	106%	16–507	0.00143
Common dolphin	16 166	24%	10 091–25 899	0.25777
Striped dolphin	NE			NE
Bottlenose dolphin	825	68%	243–2798	0.01316
Common dolphin + Striped dolphin	2116	40%	977–4584	0.03374
2011				
Minke whale	2021	56%	709–4765	0.03223
Fin whale	619	62%	1981934	0.00986
Common dolphin	32 452	27%	19 053–55 273	0.51745
Striped dolphin	4554	56%	1584–13 095	0.07262
Bottlenose dolphin	2825	68%	782–10 200	0.04505
Common dolphin + Striped dolphin	NE			NE
2012				
Minke whale	1238	62%	390–3927	0.01974
Fin whale	1389	66%	415–4646	0.02215
Common dolphin	32 521	25%	19 883–53 192	0.51854
Striped dolphin	4947	78%	1233–19 848	0.07889
Bottlenose dolphin	1191	93%	187–7575	0.01899
Common dolphin + Striped dolphin	1280	101%	235–6955	0.02040
2013				
Minke whale	2073	43%	915–4696	0.03306
Fin whale	371	67%	109–1261	0.00592
Common dolphin	53 795	26%	32 061–90 262	0.85776
Striped dolphin	13 764	75%	3622–52 310	0.21947
Bottlenose dolphin	6294	69%	1805–21 994	0.10035
Common dolphin + Striped dolphin	15 249	75%	2947–58 908	0.24314

Table 3.4. Total abundance and relative density estimated from vessel surveys covering the off-shore area between the 50 and 220 nm for species for which it was possible to estimate abundance, with correction for availability bias.

SPECIES	ABUNDANCE	% CV	95% CI	DENSITY
<i>Balaenoptera</i> spp.	906	56%	306–2682	0.00358
Fin whale	664	72%	173–2545	0.00263
Common dolphin	2406	75%	495–11 698	0.00951
Striped dolphin	20 684	50%	7726–55 375	0.08179
Bottlenose dolphin	3798	88%	756–19 089	0.01502
Atlantic spotted dolphin	5773	90%	1118–29 795	0.02283
Cuvier's beaked whale	6964	149%	709–68 396	0.02754
<i>Mesoplodon</i> spp.	871	134%	101–7545	0.00344

Cetacean distribution and abundance around Madeira

Dedicated visual systematic sea surveys were carried out in coastal waters around Madeira in 2007–2012 to study the distribution and abundance of cetacean species in the Madeira archipelago. Abundance was estimated using design-based distance sampling methods. Distribution of the density of groups and animals was modelled using GAMs. Abundance was estimated based on 69 sightings of bottlenose dolphin, 27 short-finned pilot whale, 46 Atlantic spotted dolphin, 67 short-beaked common dolphin, 22 beaked whales and 33 baleen whales, recorded during almost 9000 km of observation effort in sea conditions of Beaufort ≤ 3 . Abundance estimates are given in Table 3.5 (see Freitas *et al.*, 2014).

The abundance estimates have not been corrected for availability or perception bias and are therefore underestimates of the abundance of the species in the coastal waters of Madeira.

Table 3.5. Abundance estimates of cetaceans in coastal waters of the Madeira archipelago (4409 km²), except the Selvagens Islands.

SPECIES/FAMILY	ABUNDANCE	95% CI			CV	ANIMALS/KM ²
Bottlenose dolphin	482	365	-	607	0.14	0.11
Short-finned pilot whale	151	99	-	201	0.23	0.03
Atlantic spotted dolphin	1067	717	-	1,378	0.22	0.24
Short-beaked common dolphin	741	496	-	1,032	0.27	0.17
Balaenopteridae	20	15	-	26	0.28	0.005
Ziphiidae	27	16	-	61	0.35	0.006

The modelled distribution of bottlenose dolphin showed a greater concentration of both density of animals and density of groups in southeast Madeira, the channel between Madeira and the Desertas Islands, north and northeast of Madeira, and south-west of Porto Santo. Similarly, the short-finned pilot whale was predicted to be distributed in coastal waters of the Madeira archipelago, particularly to the southeast of Madeira Island. The spotted dolphin and the common dolphin showed a wider distribution encompassing the entire coastal sea of Madeira, with spotted dolphin

density higher throughout the south and north coasts of Madeira, while common dolphin density was higher at the edges of Madeira Island and east of the Desertas and Porto Santo Islands.

3.10 Population structure of harbour porpoise in the North and Baltic Seas

Genetic structure and morphological differences have previously suggested three distinct populations of harbour porpoises with limited overlap in distribution in the North Sea - Baltic Sea region (Wiemann *et al.*, 2010; Galatius *et al.*, 2012). Satellite telemetry data were used to determine delimitation for management purposes between the North Sea-Skagerrak populations and satellite telemetry combined with static acoustic monitoring data were used to determine delimitation between the Belt Sea and Baltic proper porpoises. The delimitation between the two former units was set as an east-west line at 56.95°N and between the latter two units a north-south line at 13.5°E (Sveegaard *et al.*, in review).

3.11 Bottlenose dolphin population structure in the Northeast Atlantic

Genetic analyses (25 microsatellites and a 682-bp portion of the mitochondrial control region, $n = 381$) have indicated that bottlenose dolphin genetic structure is hierarchical in the Northeast Atlantic. Two ecotypes, coastal and pelagic, are each further divided in two populations. In the coastal ecotype, the Coastal North (CN) population includes individuals sampled around the UK and Ireland, while the Coastal South (CS) population includes individuals of the French and Spanish coasts. The pelagic ecotype is divided into the Pelagic Atlantic (PA) and Pelagic Mediterranean (PM) populations (Louis *et al.*, 2014a). The two ecotypes have distinct ecological niches as indicated by stable isotope and stomach content analyses, but are only weakly morphologically segregated (Louis *et al.*, 2014b).

3.12 Future large-scale surveys

A North Atlantic Sightings Survey (NASS), coordinated through the North Atlantic Marine Mammal Commission (NAMMCO), will take place in summer 2015 as the latest in a series of such surveys previously conducted in 1987, 1989, 1995, 2001 and 2007. The survey will cover a large proportion of central and eastern North Atlantic waters off Norway, Iceland, the Faroe Islands and Greenland. Details have yet to be finalised but there is ongoing coordination between NASS and SCANS-III (see below) to ensure that results can be combined effectively.

A SCANS-III proposal incorporating a survey of all European Atlantic waters supported by all European Atlantic range states was submitted to the LIFE Nature programme in 2014 but was rejected at the technical stage in January 2015. The coordinators of SCANS-III are in discussion with Member States with a view to agreeing a project focusing only on the survey in summer 2016.

WGMME (2014) strongly supported the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2016 and recommended that it is supported by all range states and by ICES, the Agreement on the Conservation of Small Cetaceans in the Baltic and North Seas (ASCOBANS) and the European Commission. Continuation of these surveys is essential to the accurate estimation of absolute abundance for several species that are required for reporting under the Habitats Directive and the Marine Strategy Framework Directive (MSFD).

WGMME notes the urgent need for a new large-scale absolute abundance survey for cetaceans in European Atlantic waters, following those carried out in 1994 and 2005–2007, to fulfil EU Member State (MS) obligations under the MSFD and Habitats Directive. WGMME recommends that ICES, OSPAR and Member States support the SCANS-III initiative.

WGMME further notes the need for equivalent surveys in other European Seas covered by the MSFD where no large-scale cetacean abundance surveys have been undertaken. There is an ongoing initiative under the auspices of the Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and contiguous Atlantic area (ACCOBAMS) to carry out a cetacean survey in the Mediterranean and Black Sea. The results from such a survey would be complementary to SCANS-III and would help MS meet MSFD and Habitats Directive requirements in that region.

4 ToR b and f

ToR B: Compile a matrix of threats to the predominant cetacean species in each of the MSFD regional seas. Consider ways in which this information could be incorporated into the ICES advice.

ToR F: Produce four short paragraphs for the ICES Ecosystem Overviews on the state of sea mammal populations, one paragraph for each of the following ICES ecoregions: Greater North Sea, Celtic Seas, Bay of Biscay and the Iberian coast and Baltic Sea.

4.1 Introduction

Given the synergies between ToR B ('Compile a matrix of threats to the predominant cetacean species in each of the MSFD regional seas. Consider ways in which this information could be incorporated into the ICES advice') and ToR F ('Produce four short paragraphs for the ICES Ecosystem Overviews on the state of sea mammal populations, one paragraph for each of the following ICES ecoregions: Greater North Sea, Celtic Seas, Bay of Biscay and the Iberian coast and Baltic Sea'), it was agreed that these would be combined to avoid repetition.

Through the requirements of the Common Fisheries Policy (EU Regulation 1380/2013), during 2015, ICES will attempt to integrate issues of fishery bycatch on marine mammals into the ICES fish stock advice. The work presented under ToR B is needed to support that process by identifying regions/species more at risk of bycatch and to put bycatch into the context of other natural and anthropogenic pressures on cetacean populations. The fish stock advice also includes short summary ecosystem overviews for the region on which the advice is being given. In addition, ICES (2013a) notes that the ecosystem overviews will provide information for other aspects of ICES work, including further developments related to the Marine Strategy Framework Directive (MSFD). The contents of the ecosystem overviews will be directed at groups within ICES as well as external users of the advice and client commissions, e.g. OSPAR and the European Commission. It is expected that the overviews will provide key information on the abundance and distribution of marine mammals and consider the key anthropogenic pressures exerted upon them and this work is, therefore, closely linked to ToR B.

4.2 Development of the threat matrix

The MSFD lists four marine regions in Europe: the Baltic Sea, the Northeast Atlantic Ocean, the Mediterranean Sea and the Black Sea (MSFD, 2008). In addition, a number of subregions have been defined within the Northeast Atlantic and Mediterranean regions (Figure 4.1). Here we focus on the Baltic Sea and the subregions of the Northeast Atlantic and therefore create five matrices of threats, for the Baltic Sea, the Greater North Sea, the Celtic Seas, the Bay of Biscay and the Iberian Coast, and the Macaronesian biogeographic region. For each of these subregions, we include those marine mammal species selected for the OSPAR common indicators and/or selected by Member States (MS) in their initial evaluations for the MSFD reporting and also those species considered common and regular by previous compilation exercises (Waring *et al.*, 2009; Evans, 2010). Thus species that are rare in a region (even if common elsewhere) and vagrant species were normally excluded. We have not at this stage included Mediterranean and Black Sea and it would be desirable to co-opt ex-

perts from these regions before extending the exercise. The present summary draws on the ASCOBANS threat matrix, and evaluations of ecosystem status conducted for regional seas and individual MS evaluations under the MSFD and Habitats Directive as well as expert knowledge.



Figure 4.1. The Regional Sea divisions for MSFD assessments.

Marine mammals are subject to a range of anthropogenic pressures and threats, some of which have negative impacts ranging from increased stress and higher energetic costs, via sublethal effects on reproduction and immune function, to mortality. In more serious cases, effects may be manifest at population level (see JNCC, 2007; Hammond *et al.*, 2008; MAGRAMA, 2012). These threats include interactions with fisheries (e.g. bycatch of marine mammals in fishing gear) (Silva *et al.*, 2011; Arbelo *et al.*, 2013; ICES, 2014b), collisions with ships (e.g. Laist *et al.*, 2001; Panigada *et al.*, 2006; Evans *et al.*, 2011; Arbelo *et al.*, 2013), underwater noise (e.g. as generated by shipping or seismic activities) (e.g. Evans and Nice, 1996; Gordon *et al.*, 2003; David, 2006; Arbelo *et al.*, 2013; Jepson *et al.*, 2013), prey depletion (caused by overfishing), habitat loss or degradation, pollution (e.g. Berggren *et al.*, 1999; Bennet *et al.*, 2001; Beineke *et al.*, 2005; Davison *et al.*, 2011; Law *et al.*, 2012; Méndez-Fernández *et al.*, 2014a,b), marine debris (e.g. Laist, 1987; 1997; WCDS, 2011; ASCOBANS, 2013; Baulch and Perry, 2014; Lusher *et al.*, 2015), offshore development (including oil, gas and renewable energy, effects of which include underwater noise as well a potential habitat loss or collision risk associated with installations) (e.g. Wilson *et al.*, 2007; Bailey *et al.*, 2014), and climate change (Evans and Bjørge, 2013).

The threats/pressures listed (see Table 4.1) represent those thought to have most relevance to marine mammals and have been extracted from the list of pressures (grouped by pressure themes) agreed by the Intersessional Correspondence Group on

Biodiversity Assessment and Monitoring (ICG-COBAM, 2012). Threat levels are classified as high, medium or low (i.e. following a traffic light system), for each species-region combination, using the following criteria:

High (red) = evidence or strong likelihood of negative population effects, mediated through effects on individual mortality, health and/or reproduction;

Medium (yellow) = evidence or strong likelihood of impact at individual level on survival, health or reproduction but effect at population level is not clear;

Low (green) = possible negative impact on individuals but evidence is weak and/or occurrences are infrequent.

The category “other” (no colour) was also defined for cases where there was little or no information on the impact of these pressures on marine mammals or the threat is absent or irrelevant (in this latter case it was indicated in the corresponding cell in the table) for a particular region-species combination. Results reflect both regional differences in pressures and differences in species biology or habitat. Thus teuthophagous species are more likely to ingest plastic bags, beaked whales are particularly susceptible to mid-frequency sonar and coastal species are generally exposed to higher levels of pollutants.

Table 4.1. Descriptions of pressures by category, following ICG-COBAM (2012). In addition, a column is added where examples of the known effects of these pressures on marine mammals are given.

PRESSURE	PRESSURE DESCRIPTION	COMMENTS ON KNOWN EFFECTS ON MARINE MAMMALS
Contaminants	Increase in the level of contaminants (transition elements, hydrocarbons and <i>polycyclic aromatic hydrocarbons</i> (PAHs), Synthetic compounds, etc.)	Effects on reproduction caused by polychlorinated biphenyl (PCBs), immunosuppression leading to disease susceptibility; organ damage by heavy metals
Nutrient enrichment	Increased levels of nitrogen, phosphorus, silicon (and iron) in the marine environment.	Although environmental effects included oxygenation, algal blooms, changes in community structure of benthos and macrophytes, no negative effects have been reported for marine mammals
Habitat loss	Permanent loss of marine habitats, associated with land claim, new coastal defences, etc.	Effects likely to be only on haul-out sites for seals
Habitat degradation	Damage to marine habitats and their ecosystems	Effects mainly on coastal species
Litter (including microplastics and discarded fishing gear)	Marine litter includes: plastics, metals, timber, rope, fishing gear, etc. and their degraded components, e.g. microplastic particles	Some species, i.e. beaked whales, believed to be particularly at risk due to their mode of feeding that can make them especially vulnerable to the ingestion of marine debris

PRESSURE	PRESSURE DESCRIPTION	COMMENTS ON KNOWN EFFECTS ON MARINE MAMMALS
Sonar	Active sonar operates at medium frequencies (2–10 KHz) and has been used for a variety of purposes including in military activities	Strandings of beaked whales, possibly for other species, have been linked to the use of sonar in military exercises. Effects include hearing damage, multi-focal gas emboli, behavioural disruption
Seismic surveys	Seismic surveys use airguns to generate loud low frequency sounds (largely <100 Hz) to explore the structure underneath the seabed; seismic vessels can also emit high frequency sounds	There is no evidence of direct mortality but concern on the individual fitness and population consequences of observed displacement and change of behaviour
Pile-driving	Pile driving is considered one of the loudest sources of low frequency (<100 Hz) pulsed sounds	There is no evidence of direct mortality but concern on the individual fitness and population consequences of displacement and change of behaviour
Shipping	Noise from ships has a low frequency range, i.e. less than 1kHz, although small pleasure vessels generate higher frequency sounds due to propeller cavitation. There has been a large increase in ambient noise in recent years, particularly in the northern hemisphere	There is no evidence of direct mortality but concern on the individual fitness and population consequences of displacement and change of behaviour
Barrier to species movement (offshore windfarm, wave or tidal device arrays)	The physical obstruction of species' movements, including local movements and regional/global migrations	Significant consequences are expected to be caused by avoidance to construction/operation noise rather than direct mortality
Death or injury by collision (with ships)	Injury or mortality from collisions of biota with moving structures	Direct mortality observed in a wide range of species, particularly baleen whales and large odontocetes such as sperm whale; population consequences difficult to determine
Death or injury by collision (with tidal devices)	Injury or mortality from collisions of biota with static structures	Rotating turbines and other fixed structures could pose a collision threat, especially in rough seas
Introduction of microbial pathogens	Untreated or insufficiently treated effluent discharges and run-off from terrestrial sources and vessels	No evidence available that pathogens found in marine mammals had a human origin
Removal of target and non-target species (prey depletion)	The commercial exploitation of fish & shellfish stocks, including smaller scale harvesting, angling and scientific sampling. Ecological consequences include the sustainability of stocks, impacting energy flows through foodwebs and the size and age composition within fish stocks	Probably a major determinant of spatio-temporal variation in species distributions. Population consequences more difficult to determine

PRESSURE	PRESSURE DESCRIPTION	COMMENTS ON KNOWN EFFECTS ON MARINE MAMMALS
Removal of non-target species (bycatch)	Bycatch associated with all fishing activities. It addresses the direct removal of individuals associated with fishing/harvesting. Ecological consequences include foodweb dependencies, population dynamics of fish, marine mammals, turtles and seabirds	Evidence of poor nutritional status
Disturbance (e.g. wildlife watching)	Disturbance of biota by anthropogenic activities, e.g. increased vessel movements, increased personnel movements, increased tourism, increased vehicular movements on shore, etc.	There is no evidence of direct mortality but concern on the individual fitness and population consequences of observed displacement and change of behaviour
Deliberate killing + hunting		Hunting of marine mammals is carried out by northern OSPAR countries under the auspices of NAMMCO. Direct hunt of long-finned pilot whales and opportunistically other odontocetes in the Faroe Islands. Minke whales and seals are hunted in Norway and there has been lethal sampling of other cetaceans for research. Anecdotal reports of direct killing of dolphins for bait use and in some rare cases human consumption. Seals are killed by fishermen, both legally (under licence) and illegally.

4.3 Ecosystem overviews

The last ecosystem overviews were produced in 2008 for the Greater North Sea, Celtic Sea and West Scotland, Bay of Biscay and western Iberian waters, and the Baltic Sea (ICES, 2008). For the purposes of this ToR, these were reviewed and updated to produce new ecosystem overviews. In addition, an overview was also produced for the Macaronesia region. The key pressures in each region were identified from the 2013 Article 17 Favourable Conservation Status reports that each Member State is required to produce under the Habitats Directive (<http://bd.eionet.europa.eu/article17/reports2012/>).

4.3.1 The Baltic Sea

In the Baltic Sea (Table 4.2), three seal species and the harbour porpoise occur on a regular basis. There are an additional thirteen cetacean species and one seal species that have been noted as rare or vagrants.

Seals

The grey seal is regarded as a single population, which used to have a distribution covering the entire Baltic Sea area (Graves *et al.*, 2007). Bounty hunting campaigns (starting in 1889) and pollution with organochlorines (from the 1960s onward) caused a reduction to a few thousand individuals in the late 1970s (Boedeker *et al.*, 2002).

Reduced hunting pressure and contaminant loads have allowed the population to recover and more than 30 000 individuals were counted during the moult in 2014, a figure that does not include seals at sea during the survey. This is still lower than the estimated abundance of 80 000–100 000 before the bounty hunting campaigns (Harding and Härkönen, 1999) and grey seals have not spread to all suitable habitat in the southwestern Baltic and Kattegat where recolonization is very slow in Germany and Poland (Schwarz *et al.*, 2003; von Nordheim, 2011). Pollutants may still affect the population as well as overfishing, coastal development and bycatch.

The ringed seal is divided into two or three populations, a northern population in the Sea of Bothnia with about 9000–12 000 seals (T. Härkönen, pers. comm.), a southern population in the Gulf of Riga with 1000–1500 seals (I. Jussi, pers. comm.) and 50–100 individuals in the Gulf of Finland (M. Verevkin, pers. comm.). Population models suggest a total Baltic abundance of 180 000 to 220 000 at the beginning of the 20th century (Harding and Härkönen, 1999). The ringed seal abundance in the Bothnian Sea has increased by 4.6% annually since 1988 (T. Härkönen, pers. comm.), less than half the intrinsic capacity of 10% (Karlsson *et al.*, 2007). The estimated numbers in the Gulf of Finland have decreased from >300 to <100 recently (M. Verevkin, pers. comm.). In the Gulf of Riga, there is no sign of recovery and there are indications of recent decline (Baltic Marine Environment Protection Commission - Helsinki Commission (HELCOM) Seal Expert Group). The ringed seal is still thought to be affected by pollutants although the health status has significantly improved during the last decades (Nyman *et al.*, 2002; Routti, 2009). As an ice-breeding species, it is affected by warmer winters and less ice coverage in recent years (ICES, 2005a; Sundqvist *et al.*, 2005). Other threats include bycatch, disturbance, and increasing shipping destroying the pack ice habitat (Stenman *et al.*, 2005).

The harbour seal is found in two populations in the Baltic, one around Kalmar Sound on the Swedish east coast and one in the southwestern Baltic with haul outs in southern Sweden and Denmark (Stanley *et al.*, 1996). The Kalmar Sound population has suffered a dramatic decline from an estimated 5000 to about 200 seals in the 1970s, caused by hunting and probably impaired reproduction caused by pollutants (Härkönen and Isakson, 2010). Since the mid-1980s, the population has grown at an annual rate of ca. 9%. In 2013, 900 seals were counted during the moult, a number that does not include seals at sea during the survey (T. Härkönen, pers. comm.). In the southwestern Baltic the historical abundance is unknown, but definitely much greater than the 168 seals counted in the first systematic survey in 1988 (Olsen *et al.*, 2010). In that year and in 2002, the population was hit by the Phocine Distemper Virus epizootic. The impact of the 1988 epizootic is unknown and approximately one third of the population died in the 2002 epizootic (Olsen *et al.*, 2010). Except for this, the population has grown by a rate of around 10% since 1988 and 1563 seals were counted in 2013, a number that does not include seals at sea during the survey (A. Galatius, pers. comm.).

Cetaceans

For harbour porpoises, there is evidence of a precautionary population split between the Belt Sea and porpoises in the Baltic proper (Wiemann *et al.*, 2010; Galatius *et al.*, 2012). In the Baltic proper, a large population decline has been observed during the past 50–100 years (Skóra *et al.*, 1988; Koschinski, 2002) and this population is listed as critically endangered under the International Union for Conservation of Nature (IUCN) red list. This population was estimated at 447 (95% CI: 90–997) animals by the Static Acoustic Monitoring of the Baltic Sea Harbour Porpoise (SAMBAH) project

using static acoustic monitoring at 304 locations in the Baltic (SAMBAH, 2014). Based upon spatial monitoring, during May–December, the porpoise population in the Baltic Proper was clearly separated from that in the Western Baltic, with a major breeding area identified around the Midsjö offshore banks southeast of Öland (Sweden). In the Baltic Proper, acoustic detections were highest in the south, but the species was also detected further east and north off the coasts of Poland, Lithuania, Latvia and Finland (I. Carlén, pers. comm.).

The Belt Sea population has a much greater abundance, most recently estimated at 40 475 (95% CI: 25 614–65 041) (Viquerat *et al.*, 2014). The population trend of the latter population is uncertain (ASCOBANS 2012; Viquerat *et al.*, 2014). Major threats include bycatches, pollutants and noise.

Table 4.2. Threat matrix for the Baltic Sea.

		HARBOUR PORPOISE	GREY SEAL	HARBOUR SEAL	RINGED SEAL
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants	H	H	H	H
	Nutrient enrichment	L	L	L	L
PHYSICAL LOSS	Habitat loss				H
PHYSICAL DAMAGE	Habitat degradation	H	H	H	H
OTHER PHYSICAL PRESSURES	Litter (inc. microplastics and discarded fishing gear)	L	L	L	L
	Underwater noise changes				
	Military activity				
	Seismic surveys				
	Pile-driving	M			
	Shipping	M			
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)	L	L	L	L
	Death or injury by collision (with ships)	L	L	L	L
	Death or injury by collision (with tidal devices)	L	L	L	L
	Introduction of microbial pathogens	L	L	L	L
BIOLOGICAL PRESSURES	Removal of target and non-target species (prey depletion)	M	M	M	M
	Removal of non-target species (marine mammal bycatch)	H	M	M	H
	Disturbance (e.g. wildlife watching)	L	L	L	L
	Deliberate killing + hunting	Absent	M	M	M

4.3.2 Greater North Sea

There are two common species of pinniped in the North Sea, the grey seal (*Halichoerus grypus*) and the harbour seal (*Phoca vitulina*). Both of these species have

gone through large population changes over the past century. In addition to these residents, there are five vagrant species: ringed seal (*Phoca hispida*), bearded seal (*Erignathus barbatus*), harp seal (*Phoca groenlandica*), hooded seal (*Cystophora cristata*) and walrus (*Odobenus rosmarus*).

There are 24 species of cetacean that have been recorded in the Greater North Sea region, although few of these constitute resident representatives of the North Sea Ecosystem. There are three common species, minke whales (*Balaenoptera acutorostrata*), harbour porpoises (*Phocoena phocoena*), and white-beaked dolphins (*Lagenorhynchus albirostris*) and a small resident coastal population of bottlenose dolphins (*Tursiops truncatus*) occurring in eastern Scottish waters. Additionally, five species are considered regular but uncommon, short-beaked common dolphin (*Delphinus delphis*), Atlantic white-sided dolphin (*Lagenorhynchus acutus*), long-finned pilot whale (*Globicephala melas*), killer whale (*Orcinus orca*) and Risso's dolphin (*Grampus griseus*). Six species are rare, striped dolphin (*Stenella coeruleoalba*), sperm whale (*Physeter macrocephalus*), humpback whale (*Megaptera novaeangliae*), fin whale (*Balaenoptera physalus*), sei whale (*B. borealis*), northern bottlenose whale (*Hyperoodon ampullatus*), and nine species are vagrant, beluga (*Delphinapterus leucas*), Sowerby's beaked whale (*Mesoplodon bidens*), false killer whale (*Pseudorca crassidens*), narwhal (*Monodon monoceros*), Cuvier's beaked whale (*Ziphius cavirostris*), Blainville's beaked whale (*Mesoplodon densirostris*), Gervais' beaked whale (*M. europaeus*), North Atlantic right whale (*Eubalaena glacialis*), and Bryde's whale (*Balaenoptera edeni*).

Seals

Harbour seals come ashore in sheltered waters, typically on sandbanks and in estuaries, but also in rocky areas. They haul out regularly on land in a pattern that is often related to the tidal cycle. Harbour seals normally feed within 40–50 km around their haul out sites (SCOS, 2013). They take a wide variety of prey including sandeels, gadoids, herring and sprat, flatfish, octopus and squid. Diet varies seasonally and from region to region with approximately 3–5 kg prey being consumed per seal per day depending on the prey species (SCOS, 2013).

Because of extensive hunting, followed by reduced reproductive rates owing to effects of contamination, the populations of harbour seals along the continental coast reached an all-time low in the 1970s (SCOS, 2013). Subsequently, the harbour seal populations have increased steadily at an annual rate of 4%, although this has been affected by two major interruptions due to outbreaks of the phocine distemper virus (PDV) in 1988 and 2002 (ICES, 2008). The population along the east coast of England (mainly in The Wash) was reduced by 52% following the 1988 phocine distemper virus (PDV) epidemic, and by 22% in the second outbreak. Following this second outbreak, counts in the Wash and eastern England did not demonstrate any recovery until 2009. This is in contrast to the adjacent European colonies in the Wadden Sea, which have experienced continuous rapid growth since 2002, although growth rates are now thought to be slowing (SCOS, 2013).

Over the last 15 years, declines in the harbour seal population using the Orkney (75%), Shetland (30%), and the Scottish North Sea coasts (85% in Firth of Tay) have been recorded since 2000 (SCOS, 2013). However, such declines are not occurring at all locations. For example, the Moray Firth count declined by 50% before 2005, then has been increasing, declining or remaining stable. Similarly, the Outer Hebrides count declined by 35% between 1996 and 2008 but the 2011 count was >50% higher than the 2008 count. While the PDV epidemic affected these colonies in 1998, it did

not in 2002. The reasons for these recent declines are unknown, although thought to be different in different areas. These include natural factors such as competition with grey seals and the impact of toxins from harmful algae, as well as anthropogenic factors such as interactions with vessels (SCOS, 2014). The counted population of moulting harbour seals in the North Sea is currently close to 62 000 (including Orkney and Shetland) (SCOS, 2013).

Grey seals forage in the open sea and return regularly to haul out on land where they rest, moult and breed. Individual grey seals based at a specific haul-out site will make repeated trips to the same region to feed, which can be over 100 km away. They will occasionally move to a new haul-out site and begin foraging in a new region, for example, movements of grey seals between haul outs in the North Sea to the Outer Hebrides having been recorded (SCOS, 2013). Grey seals are generalist feeders, foraging mainly on the seabed at depths of up to 100 m although they are probably capable of feeding at all the depths found across the European continental shelf. They take a wide variety of prey including sandeels, gadoids (cod, whiting, haddock, ling), and flatfish (plaice, sole, flounder, dab). Amongst these, sandeels are typically the predominant prey species. Diet varies seasonally and from region to region. Food requirements depend on the size of the seal and fat content (oiliness) of the prey, but an average consumption estimate is 4 to 7 kg per seal per day depending on the prey species (SCOS, 2013).

Grey seals occur predominantly along the British coasts of the North Sea and have been increasing at an annual rate of up to 10% (SCOS, 2013). In recent years, new colonies have been founded as population growth continues, although there are recent signs that the rate of growth is levelling off in some colonies (SCOS, 2014). The pup production in the North Sea including Orkney is currently close to 32 500, with approximately 98% occurring in the UK (SCOS, 2014).

In the past, interactions with commercial fishing have been considered to be an important anthropogenic pressure (ICES, 2008). Currently, key anthropogenic pressures on seals in the Greater North Sea region are pollutants, disturbance while on land, and interactions with vessels. The occurrence of 'corkscrew' deaths has attracted considerable attention in the last few years, with attention initially focused on ducted propellers although more recently at least some such deaths have been attributed to predation by grey seals (van Neer *et al.*, 2014, 2015; Thompson *et al.*, 2015; see also ToR c).

Harbour seal conservation status was considered to be unfavourable to bad in the collated Member State Article 17 reports for the Marine Atlantic region (see <http://bd.eionet.europa.eu/article17/reports2012/>). The area actually covers the Celtic Seas and the Bay of Biscay and Iberian Peninsula. The range and habitats of the species were largely considered to be in a favourable condition, but the UK report of major declines in some colonies lead to the overall conclusion of unfavourable-bad. In complete contrast, the conservation status of grey seals is considered to be favourable by all Member States in the 2013 reporting. It can therefore be concluded that the human pressures exerted upon grey seals within the North Sea region are not having a significant impact at the population level for this species.

Cetaceans

The population estimates derived from surveys carried out in 2005 are 324 000 (CV=0.22; 95% CI=256 300–549 700) for harbour porpoise, 10 700 (CV=0.38; 95% CI=9200–29 600) for white-beaked dolphin and 13 734 (CV=0.41; 95% CI=9800–36 700)

for minke whale (Hammond *et al.*, 2013). Comparisons to an earlier survey in 1994 (Hammond *et al.*, 2002), indicate that harbour porpoise have shifted their focal distribution from the northern part of the North Sea to the southern part (Hammond *et al.*, 2013). This redistribution is thought to be related to the availability of the prey resource. Minke whales and white-beaked dolphins have overlapping distributions and are mainly found in the northern part of the North Sea, with no obvious changes in distribution between the two surveys (Hammond *et al.*, 2013). A third large-scale abundance survey is being planned for 2016 (see ToR a). Cheney *et al.* (2013) provides the most recent population estimate of 195 (95% HPDI: 162–253) for the east coast of Scotland bottlenose dolphin population, which is considered to be stable or increasing (Cheney *et al.*, 2012).

The collated Member State Article 17 reports required under the Habitats Directive (see <http://bd.eionet.europa.eu/article17/reports2012/>) indicate that harbour porpoise, minke whale and white-beaked dolphin are all considered to be in a favourable conservation status. The coastal bottlenose dolphin population utilising the Scottish northeast coast is also considered to be in favourable condition (JNCC, 2013).

Key anthropogenic pressures are bycatch in static nets for harbour porpoise (Bjørge *et al.*, 2013; ASCOBANS, 2014; ICES, 2014a; and see Table 4.3) and potential entanglement of minke whale in mooring ropes and creel lines (Northridge *et al.*, 2010). For bottlenose dolphin, direct and indirect disturbance and harassment (e.g. boat traffic and commercial dolphin watching and interactive programs) are considered an issue (Pirodda *et al.*, 2015). In general, activities such as marine construction (e.g. pile driving), demolition and activities such as geophysical surveys create noise and can lead to changes in distribution (e.g. ICES, 2010; Bailey *et al.*, 2010; Brandt *et al.*, 2011; Scheidat *et al.*, 2011; Dahne *et al.*, 2013). Although concerns with respect to pollutants such as persistent organic pollutants (POPs) (including PCBs) leading to immunosuppression have been raised (e.g. Jepson *et al.*, 1999; 2005; ICES, 2010; Yap *et al.*, 2012), issues associated with reproductive failure in harbour porpoise (Murphy *et al.*, accepted) and bottlenose dolphin (Jepson *et al.*, submitted) have implications for future conservation status assessments.

Table 4.3. Threat matrix for the greater North Sea.

[illegible]

		HARBOUR PORPOISE	COMMON DOLPHIN	WHITE- BEAKED DOLPHIN	ATLANTIC WHITE- SIDED DOLPHIN	RISSO'S DOLPHIN	MINKE WHALE	LONG- FINNED PILOT WHALE	KILLER WHALE	COASTAL BOTTLENOSE DOLPHIN	GREY SEAL	HARBOUR SEAL
Death or injury by collision	Death or injury by collision (with ships)	M	M	L	L	L	M	L	L	M	L	L
	Death or injury by collision (with tidal devices)	Risk of collision leading to death or injury is considered possible, but no evidence of such an occurrence to date										
BIOLOGICAL PRESSURES	Introduction of microbial pathogens	L	L	L	L	L	L	L	L	L	L	L
	Removal of target and non-target species (prey depletion)	M	L	L	L	L	M	L		L	L	L
	Removal of non-target species (marine mammal bycatch)	H	L	L	L	L	M	L	L	L	M	M
	Disturbance (e.g. wildlife watching)	L	L	L	L	L	L	L	L	M	L	M
	Deliberate killing + hunting	Does not occur						L	Does not occur		M	M

4.3.3 Celtic Seas including West Scotland

Thirty species of cetaceans have been recorded in this Advisory Region, of which twelve species are regularly observed (Reid *et al.*, 2003; Evans, 2008; Wall, 2010; Baines and Evans, 2012). Minke whale is found throughout the region, particularly off western Scotland and Ireland, and in the deeper waters of the Irish Sea. Dedicated surveys (e.g. SCANS-II, *Cetacean Offshore Distribution and Abundance* in the European Atlantic (CODA) and observer programmes on ships of opportunity have recorded that bottlenose dolphins occur in large numbers offshore in the Southwest Approaches to the British Isles, and off southwestern and western Ireland, and in smaller numbers elsewhere throughout the region, with more localised populations of between 100 and 300 individuals in the Shannon Estuary (Berrow *et al.*, 2010) and in west and north of Wales (Feingold and Evans, 2014). Short-beaked common dolphin are widely distributed in shelf waters, but especially in the Celtic Sea and adjacent areas. White-beaked dolphin and Atlantic white-sided dolphin occur over the northern part of the region, the former mainly on the shelf area, and the latter mainly along and beyond the shelf edge. Harbour porpoise is the smallest, but by far the most numerous, of the cetaceans found in the Celtic Seas ecoregion, particularly southwest of Ireland, west of Scotland and parts of the Irish Sea (Baines and Evans, 2012; Wall, 2013; Hammond *et al.*, 2013). Other regular and common species include long-finned pilot whale and fin whale, with Risso's dolphin, killer whale, northern bottlenose whale, and sperm whale regular but uncommon (and occurring mainly along or beyond the continental shelf edge).

Grey seals are common in many parts of the area. Population estimates (from 2014) are available only for West Scotland, with 7900 (95% credible interval from a Bayesian model: 6000–9400) in the Inner Hebrides, and 27 000 (95% credible interval from a Bayesian model: 22 100–32 500) in the Outer Hebrides (Thomas, 2014; SCOS, 2014). The Outer Hebridean population has been stable since 1990 and the Inner Hebridean population since 2000. Equivalent pup production estimates are 4088 for the Inner Hebrides and 14 136 for the Outer Hebrides (Duck and Morris, 2014; SCOS, 2014). The majority of individuals are found in the Hebrides and in Orkney, although smaller numbers occur in Wales and around Ireland. For those latter regions, only pup production estimates exist. In Wales, pup production was estimated at 1650 from indicator sites in 2004–2005, inflated by a multiplier from 1994 synoptic surveys (Strong *et al.*, 2006). For Northern Ireland, pup production for 2010 was estimated at 100, and for the Republic of Ireland (2005) at 1600 (Ó'Cadhla *et al.*, 2007).

Studies of grey seal diet in the Western Isles from 2010 (no summer information) indicate it is dominated by sandeel and large gadids, particularly cod and ling. Other components of the diet included *Trisopterus* spp., dragonet, wrasse and cephalopods (Hammond and Wilson, 2015). There have been no studies of grey seal diet in the western Irish Sea since Kiely *et al.* (2000) identified the predominant prey species to be Norway pout, bib, poor cod, whiting, and plaice, or in the southeastern Irish Sea since Strong (1996) identified gadids, mainly whiting and *Trisopterus* spp., and flat-fish, mainly sole, were the predominant prey species.

Harbour seals are also widespread in the northern part of the ecoregion with moult counts of 834 in SW Scotland (2007), 11 057 in the Inner Hebrides and west coast (2007–2013), and 2739 in the Outer Hebrides (2011), yielding a total of 12 170 for all of West Scotland (Duck *et al.*, 2014; SCOS, 2014). Equivalent moult counts are 948 in Northern Ireland (2011), and 3489 in the Republic of Ireland (2011–2012), yielding a total of 4437 for the whole of Ireland (2011–2012) (Duck and Morris, 2012; 2013a, b).

For approximate estimates of population size, the moult counts should be multiplied by 1.39 in order to account for animals not ashore during counts.

Harbour seal diet has been studied in West Scotland (Wilson and Hammond, 2015a). In the Outer Hebrides: diet (summer) was fairly evenly spread across *Trisopterus* spp., large gadids, sea scorpions, pelagic prey (especially mackerel), and sandeels. In the Inner Hebrides and on the west coast of Scotland, it was dominated by large gadids with seasonal pulses of pelagic prey (mackerel and herring). Other components of the diet included *Trisopterus* spp. and dragonet. A comparison with previous studies showed that the diet appears not to have changed much in this region.

Cetacean bycatch in fisheries (particularly set gillnets and trawls) has been acknowledged to be a threat to the conservation of cetaceans in this ecoregion (CEC, 2002; ICES, 2010b, c, 2011a). As in other areas this mainly affects small cetaceans, notably porpoises, common and striped dolphins (ICES, 2014a), although baleen whales such as minke and humpback whale can also be affected by entanglement in creel lines and ghost netting (Northridge *et al.*, 2010). Other species that have been caught in the region include Atlantic white-sided dolphin, white-beaked dolphin, bottlenose dolphin, and long-finned pilot whale (ICES, 2014a, b). The overall scale of cetacean bycatch in this area remains unclear due to incomplete observer coverage and an absence of detailed fishing effort data in certain key fleets. Existing observer data and derived estimates of cetacean bycatch in specific fisheries in the region have not identified any major conservation concern, but there are as yet no reliable estimates of overall take (ICES, 2014a). During the 1990s in particular, the bass fishery in the western Channel, mackerel and horse mackerel trawling off SW Ireland, and gillnetting for hake in the Celtic Sea were reported as causing bycatch of porpoises and common dolphins (Tregenza *et al.*, 1997a, b; Tregenza and Collet, 1998; Ross and Isaacs, 2004). Recent management measures have eliminated one of the main sources of common dolphin bycatch in the winter, through the closure of the French and UK bass surface trawl fishery (ICES, 2014a). The obligation on netting vessels of 12 m or more from EU Member States to use pingers under regulation 812/2004 should have reduced bycatch of porpoises and probably that of common dolphins too in such fisheries, but there is little independent verification of this assumption. Limited observer coverage of the remaining fisheries makes it difficult to determine bycatch levels in the remainder of the fleets operating in the region (ICES, 2014a).

Other potential threats known to impact upon marine mammals from human activities occurring in the region include noise from seismic exploration, military activity, shipping, and pile driving; physical damage from vessel strikes; and disturbance from recreational activities.

Table 4.4. Threat matrix for Celtic Seas including West Scotland.

[illegible]

			HARBOUR PORPOISE	COMMON DOLPHIN	WHITE- BEAKED DOLPHIN	ATLANTIC WHITE- SIDED DOLPHIN	RISSO'S DOLPHIN	MINKE WHALE	LONG- FINNED PILOT WHALE	KILLER WHALE	FIN WHALE	SPERM WHALE	OFFSHORE BOTTLENOSE DOLPHIN	COASTAL BOTTLENOSE DOLPHIN	NORTHERN BOTTLENOSE WHALE	GREY SEAL	HARBOUR SEAL
	Death or injury by collision	With ships	M	M	L	L	L	M	L	L	M	M	L	M	L	L	L
		With tidal devices	Risk of collision leading to death or injury is considered possible (e.g. at Strangford Lough), but no evidence of such an occurrence to date														
BIOLOGICAL PRESSURES	Introduction of microbial pathogens		L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of target and non-target species (prey depletion)		M	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of non-target species (marine mammal bycatch)		H	H	L	M	M	M	L	L	L	L	L	L	L	L	L
	Disturbance (e.g. wildlife watching)		L	L	L	L	L	L	L	L	L	L	L	M	L	L	M
	Deliberate killing + hunting		Does not occur														M

4.3.4 The Bay of Biscay and the Iberian Peninsula

In the Bay of Biscay and Iberian Atlantic, a total of 28 species of cetacean have been recorded, but most of these must be considered vagrants and uncommon. There are six common species. SCANS-II provide abundance estimates for four of these: harbour porpoise (2400 animals, CV = 0.92), short-beaked common dolphin (18 400 animals, CV = 0.545), striped dolphin and bottlenose dolphin (5400 animals, CV = 0.69). No abundance estimates are available for long-finned pilot whale or fin whale.

There are no common species of seal occurring in this region. Grey seals are considered to be uncommon, harbour seals rare, and hooded, bearded, harp, walrus and the Mediterranean monk seal (*Monachus monachus*) are all vagrants to the region.

Key threats in this region are largely similar to those from other biogeographic regions and include bycatch, disturbance as a result of marine construction including renewable energy developments, geophysical surveys and shipping, the effect of pollutants and a loss of habitat and/or prey resources. The Bay of Biscay was identified as an area with a high risk to fin whale and sperm whale from vessel strike, since those species and shipping co-occur here at highest densities (Evans *et al.*, 2011). Based on the assessments of conservation status for the countries comprising this region (France, Spain and Portugal), the conservation status of harbour porpoise and common dolphin is unfavourable across the entire region in terms of population parameters, range and habitat. The striped dolphin is listed as unfavourable in Portugal and unknown in France and Spain, whereas bottlenose dolphin status is unknown in Spain, unfavourable in France, and favourable in Portugal. Finally, the conservation status of long-finned pilot and minke whales is unknown across the entire region.

Table 4.5. Threat matrix for The Bay of Biscay and the Iberian Peninsula.

		HARBOUR PORPOISE	COMMON DOLPHIN	STRIPED DOLPHIN	CUVIER'S BEAKED WHALE	RISSE'S DOLPHIN	LONG- FINNED PILOT WHALE	KILLER WHALE	FIN WHALE	SPERM WHALE	NORTHERN BOTTLENOSE WHALE	SOWERBY'S BEAKED WHALE	OFFSHORE BOTTLENOSE DOLPHIN	COASTAL BOTTLENOSE DOLPHIN
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants	H	M	M	L	L	M	H	L	L	L	L	L	H
	Nutrient enrichment	L	L	L	L	L	L	L	L	L	L	L	L	L
PHYSICAL LOSS	Habitat loss	L	L	L	L	L	L	L	L	L	L	L	L	L
PHYSICAL DAMAGE	Habitat degradation	L	L	L	L	L	L	L	L	L	L	L	L	L
OTHER PHYSICAL PRESSURES	Litter (including microplastics and discarded fishing gear)	L	L	L	M	L	L	L	L	L	M	M	L	L
	Sonar	L	L	L	H	L	L	L	L	L	M	M	L	L
	Seismic surveys	M	M	M	M	M	M	L	M	M	M	M	M	M
	Pile-driving	No current activity but potentially harmful												
	Shipping	L	L	L	L	L	L	L	M	L	L	L	L	L

		HARBOUR PORPOISE	COMMON DOLPHIN	STRIPED DOLPHIN	CUVIER ´S BEAKED WHALE	RISSO'S DOLPHIN	LONG- FINNED PILOT WHALE	KILLER WHALE	FIN WHALE	SPERM WHALE	NORTHERN BOTTLENOSE WHALE	SOWERBY'S BEAKED WHALE	OFFSHORE BOTTLENOSE DOLPHIN	COASTAL BOTTLENOSE DOLPHIN	
BIOLOGICAL PRESSURES	Barrier to species movement (offshore windfarm, wave or tidal device arrays)	L	L	L	L	L	L	L	L	L	L	L	L	L	
	Death or injury by collision	With ships	L	L	L	L	L	L	L	H	H	L	L	L	L
		With tidal devices	No current activity but potentially harmful												
	Introduction of microbial pathogens	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of target and non-target species (prey depletion)	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of non-target species (marine mammal bycatch)	H	H	M	L	L	L	L	L	L	L	L	M	H	
	Disturbance (e.g. wildlife watching)	L	L	L	L	L	L	L	L	L	L	L	L	M	
	Deliberate killing + hunting	Does not occur	L	Does not occur											

4.3.5 Macaronesia

The Macaronesian biogeographic region comprises four distinct island archipelagos of which three are being treated here: the Azores, Madeira, and the Canaries.

The Azores

There are 27 species recorded for the Azores, although most of these are uncommon, rare or vagrant. The common species are short-beaked common dolphin, Atlantic spotted dolphin (*Stenella frontalis*), bottlenose dolphin, short-finned pilot whale (*Globicephala macrorhynchus*), Risso's dolphin, striped dolphin, and sperm whale. Among the seals, only the harbour seal has been recorded as a vagrant in this island group.

Groups of female and juvenile sperm whales are found year-round in the waters surrounding the archipelago, attended seasonally by itinerant males (Steiner *et al.*, 2012). Matthews *et al.* (2001) estimated between 400 and 2200 females and juveniles in waters off the central summer in spring and summer from 1988 to 1994. The archipelago seems to be a calving and mating ground, with maximum breeding activity during April and June (Steiner *et al.*, 2012; Hartman *et al.*, 2014). Risso's dolphins and common dolphins are found in coastal as well as offshore waters (Hartman *et al.*, 2014). Common dolphins are present year-round, although their presence declines significantly between June and November, coinciding with the occurrence of Atlantic spotted dolphins. Bottlenose dolphins and Risso's dolphins occur primarily in near shore waters. Both species are observed year-round but encounter rates vary greatly between months.

The main threat posed to cetaceans in the archipelago of the Azores is thought to be disturbance by commercial whale-watching operators. Magalhães *et al.* (2002) and Visser *et al.* (2011) described short-term behaviour changes of sperm whales and Risso's dolphin, respectively, in the presence of whale watching boats.

Madeira

There are 29 cetacean species recorded from the Madeira Island group of which eight are considered common: bottlenose dolphin, Atlantic spotted dolphin, short-beaked common dolphin, fin whale, sperm whale, Bryde's whale, short-finned pilot whale, and striped dolphin. The Mediterranean monk seal is considered to be rare in the Madeira island group, and harbour seal a vagrant species.

Bottlenose dolphins have a preference for shallow waters, with a larger presence in the east of Madeira and south of Porto Santo, where abundance has been estimated at 482 animals (95% CI: 365–607; CV=0.14) [estimates have not been corrected for $g(0)<1$] (Freitas *et al.*, 2013). Short-finned pilot whales occur preferentially in the southeast area of Madeira, in deeper waters, with a large degree of variability of site fidelity, including residents, regular visitors and transients (Alves *et al.*, 2013). The abundance of the island-associated whales has been estimated to be around 140 animals (95%CI: 131–151) (Alves *et al.*, 2015). Common dolphins and Atlantic spotted dolphins are distributed throughout the archipelago, although the former can be found mostly in waters off Madeira, and in the east of Desertas Islands and Porto Santo, and the latter off the south coast of Madeira (Freitas *et al.*, 2014). Sperm whales, found mostly in groups, are present year-round in the archipelago (Freitas *et al.*, 2014). The presence of Bryde's whales is regular between June and November, showing seasonal site-fidelity (Freitas *et al.*, 2014). Striped dolphins are regularly observed within the archipelago, mainly around Porto Santo and the Selvagens, although the species is less

abundant than common dolphin or Atlantic spotted dolphin (Freitas *et al.*, 2012, 2014).

The main threats posed to cetaceans in this archipelago are whale-watching and marine traffic (Freitas *et al.*, 2014). The monk seal is strictly protected in Madeira, the main threat arising from fishery interactions (e.g. Karamanlidis *et al.*, 2003; Pires *et al.*, 2008).

Canaries

The Canaries have 31 recorded species, of which ten are common: bottlenose dolphin, short-finned pilot whale, Atlantic spotted dolphin, rough-toothed dolphin (*Steno bredanensis*), Risso's dolphin, sperm whale, short-beaked common dolphin, striped dolphin, Bryde's whale, and Cuvier's beaked whale. The Mediterranean monk seal is considered to be rare in the Canaries, and harbour seal a vagrant species.

Bottlenose dolphins are present throughout the year in the waters off the Canaries, closely linked to certain coastal areas of some islands, in water depths not exceeding 600 m depth (MAGRAMA, 2012). Tobeña *et al.* (2014) showed that this cetacean species undertakes regular, long-distance movements between islands. Short-finned pilot whales are found year-round throughout the archipelago. The most important area of concentration is located south of Tenerife, where the species can be found in water depths between 700 and 2000 m, and showing different degrees of site fidelity (MAGRAMA, 2012). Sperm whales can be found year-round and are distributed throughout the archipelago. Areas of high concentration have been described in the channel between Tenerife and Gran Canaria, north of La Palma, east and west of the channel between Fuerteventura and Lanzarote, and in the vicinity of the Concepcion seamount (Fais, 2015). Sperm whale abundance in offshore waters of the archipelago has been estimated around 220 animals (95% CI: 117–413, CV = 0.32) (Fais, 2015). Striped dolphin, Atlantic spotted dolphin, Risso's dolphins, and rough-toothed dolphin are present year-round in the Canaries and are distributed throughout the archipelago, although the last species appears to be relatively common around La Gomera (Ritter, 2001).

The main threats observed in waters off the Canary Islands are bycatch, mass strandings related to sonar exercises, and ship collisions (Arbelo *et al.*, 2013), as well as disturbance. The cetacean species that most often are recorded entangled in fishing gear include bottlenose, striped, common and Atlantic spotted dolphins (Arbelo *et al.*, 2013). Notable mass strandings related to sonar exercises occurred in 2002 and 2004. In 2002, a total of 14 beaked whales, including Cuvier's, Blainville's and Gervais' beaked whales, stranded with signs of haemorrhages and both gas and fat emboli some hours after an international naval exercise (Fernández *et al.*, 2005), and, in 2004, four Cuvier's beaked whales were found stranded directly after naval exercises (Fernández *et al.*, 2012). In the Canary Islands, high densities of whales and dolphins coincide with areas with relatively high levels of maritime traffic (Ritter, 2010). Sperm whales, short-finned pilot whales, Cuvier's beaked whales, fin and Bryde's whales comprise the majority of stranding records resulting from ship strikes in this archipelago (Carrillo and Ritter, 2010). Furthermore, pilot whales off southern Tenerife suffer high pressure from whale-watching vessels.

Table 4.6. Threat matrix for Macaronesia.

		SHORT-FINNED PILOT WHALE	STRIPED DOLPHIN	SPERM WHALE	RISSO ´S DOLPHIN	CUVIER ´S BEAKED WHALE	BLAINVILLE'S BEAKED WHALE	OFFSHORE BOTTLENOSE DOLPHIN	COMMON DOLPHIN	FIN WHALE	ROUGH-TOOTHED DOLPHIN	BRYDE'S WHALE	ATLANTIC SPOTTED DOLPHIN	MONK SEAL	
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants	L	M	L	L	M	M	L	M	L	M		M	M	
	Nutrient enrichment	L	L	L	L	L	L	L	L	L	L		L	L	
PHYSICAL LOSS	Habitat loss	L	L	L	L	L	L	L	L	L	L		L	L	
PHYSICAL DAMAGE	Habitat degradation	L	L	L	L	L	L	L	L	L	L		L	M/H	
OTHER PHYSICAL PRESSURES	Litter (inc. microplastics and discarded fishing gear)	L	L	L	M	M	L	L	L	L	L		L	M	
	Underwater noise changes	Military activity	L	L	L	L	H	H	L	L	L	L		L	
		Seismic surveys	M	M	M	M	M	M	M	M	M	M	M	M	M
		Pile-driving	No current activity but potentially harmful												
		Shipping	M	M	M	M	M	M	M	M	M	M	M	M	M
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)	No current activity but potentially harmful													

			SHORT-FINNED PILOT WHALE	STRIPED DOLPHIN	SPERM WHALE	RISSO ´S DOLPHIN	CUVIER ´S BEAKED WHALE	BLAINVILLE'S BEAKED WHALE	OFFSHORE BOTTLENOSE DOLPHIN	COMMON DOLPHIN	FIN WHALE	ROUGH-TOOTHED DOLPHIN	BRYDE'S WHALE	ATLANTIC SPOTTED DOLPHIN	MONK SEAL
	Death or injury by collision	With ships	M	L	M	L	M	L	L	L	M	L	L	L	L
		With tidal devices	No current activity but potentially harmful												
BIOLOGICAL PRESSURES	Introduction of microbial pathogens		L	L	L	L	L	L	L	L	L	L		L	L
	Removal of target and non-target species (prey depletion)		L	L	L	L	L	L	L	L	L	L		L	L
	Removal of non-target species (marine mammal bycatch)		L	M	L	L	L	L	M	M	L	L		M	M
	Disturbance (e.g. wildlife watching)		M	L	L	M			L	L	L	L		L	H
	Deliberate killing + hunting		Does not occur in the area												

5 ToR c: Review North Atlantic information on negative and positive ecological interactions between grey seal (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) populations

5.1 Introduction

There is increasing evidence of possible ecological interactions between harbour and grey seals in regions where their distributions overlap. In general, the effects appear to be one-sided; grey seals may have both direct (e.g. predation) and indirect (e.g. competition for prey and space) negative effects on harbour seals but there is no evidence of the reverse. The extent to which grey seal interactions with harbour seals actually impact or regulate harbour seal populations is uncertain. The one-sidedness of the interactions may in part be related to the bigger size of grey seals, and also to grey seals having been more depleted than harbour seals by hunting and thus re-establishing a balance between the two species during its recovery. The phenomenon of interactions is recent, and is at least partly a consequence of the increase of grey seal populations on both sides of the Atlantic. We here review population trends in abundance of grey and harbour seals where their distributions overlap and summarise four main ecological interactions: at-sea habitat usage, competition for prey resources, haul-out site usage, and predation of harbour seals and harbour porpoises by grey seals.

5.2 Population trends in abundance

5.2.1 Wadden Sea

Along continental Europe (except Norway) grey seals went extinct in the late middle ages (Härkönen *et al.*, 2007). Except for occasional occurrences, the species was absent from the Wadden Sea until the 1950s when observations became more frequent. The first breeding colony with only a few pups born annually was established in the Dutch Wadden Sea in the 1980s (Reijnders *et al.*, 1995), and in winter 2003/2004 at least 150 pups were born (Reijnders and Brasseur, 2003). This number has increased since then and in 2014, more than 600 pups were counted in the Wadden Sea, and more than 4200 adults were counted in the moulting season (Brasseur *et al.*, 2014). The observed increase in grey seal numbers cannot be explained by internal recruitment alone; at least one third of the one-year-old seals in the Dutch Wadden Sea will have come from the UK to eventually settle as breeding animals (Brasseur *et al.*, 2014).

Harbour seal abundance in the Wadden Sea was reduced to less than 5000 individuals in the mid-1970s because of hunting, disturbance and pollution (Reijnders *et al.*, 2010). Since then the population has been increasing, interrupted by two outbreaks of Phocine Distemper Virus (PDV) in 1988 and 2002, which killed approximately 50% of the harbour seals on both occasions (Härkönen *et al.*, 2006). In 2014, the abundance was estimated at 39 100. Thus, competition between the two species may occur in the Wadden Sea, but the return of the grey seal has not prevented the continued increase in abundance of the harbour seals in the area. It should be noted that in the Wadden Sea, grey seal abundance is still low compared to harbour seal abundance.

5.2.2 Baltic Sea and Kattegat

During the late 19th and early 20th century, bounties for shot seals were instigated in Denmark, Sweden and Finland, as seals were regarded as competition for the fisher-

ies. This led to the abandoning of traditional haul-out sites (Hult, 1943), particularly in the southern Baltic, where grey seals were largely depleted and breeding colonies disappeared in Denmark and southern Sweden by 1900 and before 1910 in Poland and Germany (Søndergaard *et al.*, 1976; Gill, 1978; Härkönen *et al.*, 2007). In the latter half of the 20th century, pollution with persistent organochlorines led to massive reproductive failure in Baltic seals, and it is estimated that the abundance of the Baltic grey seals dropped from almost 100 000 in the early 20th century to approximately 4000 in the 1970s (Harding and Härkönen, 1999). In the mid-1970s, grey seal abundance commenced a slow recovery to an estimated 5500 individuals in 1996 (Harding and Härkönen, 1999). During the 1980s and 1990s decreasing levels of organochlorines have led to improved health conditions (Bergman, 1999) and more than 32 000 were counted in 2014. During this recovery, the distribution of Baltic grey seals has expanded southwards, bringing them into contact with the harbour seals occurring in the southwestern Baltic and Kattegat.

Harbour seals in the southwestern Baltic and Kattegat were subjected to the same bounty programmes as the grey seals and suffered large declines in abundance (Søndergaard *et al.*, 1976). Protection measures were taken in the 1960s and 1970s after which the abundances increased from a few thousand (Olsen *et al.*, 2010), interrupted by the two outbreaks of PDV (Härkönen *et al.*, 2006). The harbour seals in the Baltic and Kattegat are divided into three population units, Kalmar Sound, Southwestern Baltic and Kattegat (Olsen *et al.*, 2014). All these units have been increasing in abundance during the return of the grey seals to the respective areas (Härkönen *et al.*, 2013; Sveegaard *et al.*, 2014). The return of the grey seal has not prevented the continued population growth of these populations. However, there are indications of a shift in the use of haul-out sites by harbour seals in the southwestern Baltic, away from the sites favoured by the grey seals (Mikkelsen *et al.*, in prep).

5.2.3 France

Both grey and harbour seals in France are at the southern limit of their range in the Northeast Atlantic. Seal numbers are low compared to those reported around the British Isles, but they are increasing for both species, and contrasting dynamics in grey seal numbers were recently observed between the eastern and the western end of the Channel. Until recently, the three harbour seal colonies were located in the eastern part of the Channel (and the *baie du Mont Saint-Michel*) while the two main grey seal colonies were located further west in Brittany. During the last years, grey seals have rapidly increased in the eastern Channel, using the same haul-out sites as the harbour seals, especially in the *baie de Somme* (main harbour seal colony in France with 80 pups born in 2014). A maximum of 285 grey seals were censused on haul-out sites during summer 2014 in the eastern Channel, while harbour seal numbers peaked at 470 individuals during the same summer period. In *baie de Somme*, harbour seal numbers are still increasing at a rate of +17% per year while grey seals increase at +24% per year. In Normandy harbour seal pup production was 44 pups in 2014 and the maximum number of seals on haul-out sites was of 215 individuals during the moulting season; these numbers increased by 13% per year. Grey seals in Brittany also increased, but at a lower rate: a maximum number of 285 grey seals were censused during the moulting season, and the pup production is estimated at 45 pups minimum for 2014. Rates of increase were estimated at 4.5% and 5.8% per year respectively in the Iroise Sea and the Sept Iles archipelago. These rates of increase contrast to those reported further east, and recent telemetry studies conducted on grey seals in both areas showed contrasting movement patterns: grey seals tracked from

western Brittany mainly moved towards the southwest British Isles and Ireland while those tracked from the *baie de Somme* moved into the southern North sea and eastern Channel (Figure 4.5 in ICES, 2013b). These results appear to be consistent with the recognition of two management units between (1) North Sea (OSPAR Region II) and (2) western Britain, Ireland, and western France (OSPAR Region III and part of Region IV, OSPAR, 2014).

5.2.4 UK and Ireland

Since the 1988 PDV outbreak, harbour seal moult counts in some regions of the UK have increased or remained stable while dramatic declines in several other regions have been documented after 2000 (Figure 5.1). The number of animals counted during aerial surveys in Orkney, Shetland and the Southeast coast of Scotland continue to decline at unprecedented rates. In Orkney, the mean annual rate of decline between 2001 and 2010 was 13% (Lonergan *et al.*, 2013; 95% CI: 10.8–14.8) and the population in the Firth of Tay in Scotland has declined by 93% since 2000 (SCOS, 2014). In contrast, grey seal pup production and modelled total population size has been increasing exponentially in the North Sea and Orkney regions (Figure 5.2; SCOS, 2014), leading to hypotheses about a concomitant increase in interspecific competition between the two species.

However, in other regions harbour seal population trends are stable (e.g. west coast of Scotland), or even increasing (e.g. southeast England, southwest Ireland) while grey seal numbers have stabilised (Figures 5.1 and 5.2; SCOS, 2014). There is little information with which to assess contrasting local population trajectories in Ireland. There was an approximately 15% increase in minimum harbour seal population size between 2003 and 2011/2012, and between 2005 and 2009/2012 there was an approximately 25% increase in the grey seal breeding population size estimate (M. Cronin, pers. comm.).

5.2.5 Western North Atlantic

The population of harbour seals on Sable Island, Canada, declined rapidly through the 1990s. Maximum pup production was estimated to be ~ 600 in 1989 but declined to less than twelve by 2002. This was followed by a decline in older age classes in subsequent years (Hammill *et al.*, 2010). During the 1990s, the sympatric grey seal population increased at a rate of 13% per year and by 2007 Sable Island was home to the largest grey seal breeding colony in the world. In 2014, the estimated total grey seal population size on Sable Island was 394,000 (95% CI: 329 000 to 682 000) (DFO, 2014).

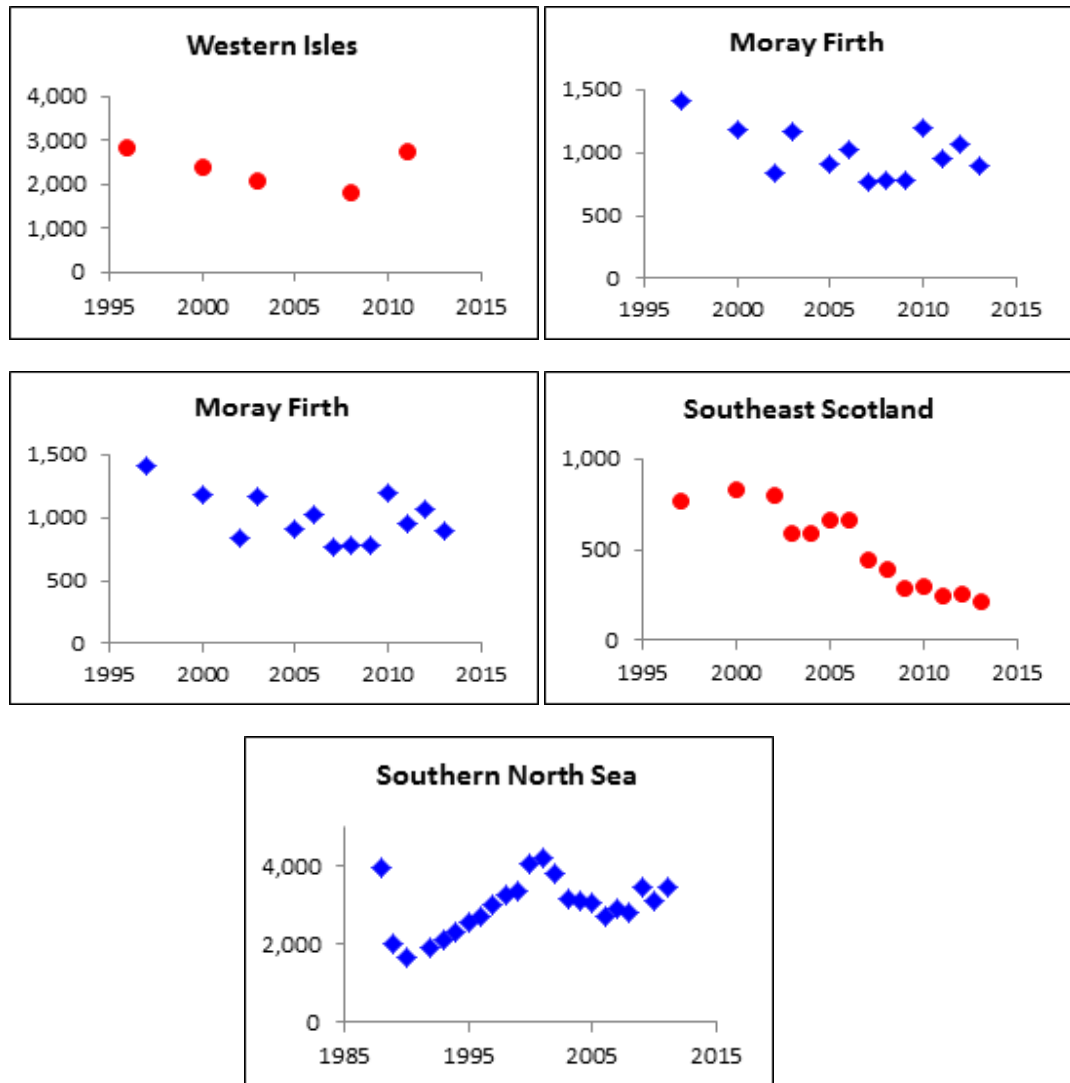


Figure 5.1. Trends in harbour seal August moult counts in regions around the UK (Data: Sea Mammal Research Unit).

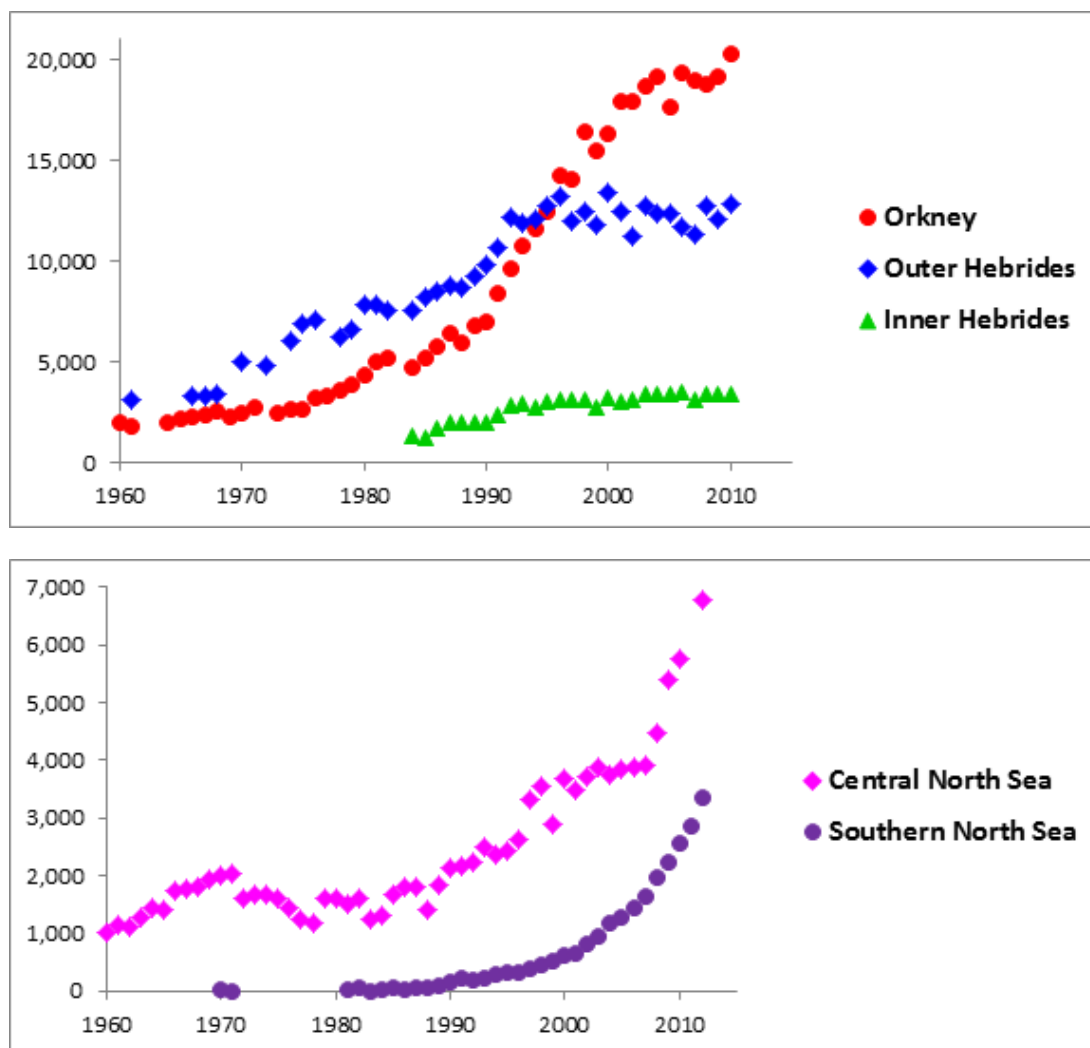


Figure 5.2. Trends in grey seal pup production at major breeding colonies in the UK (Data: Sea Mammal Research Unit).

Prior to the early 1980s, grey seals were a rare species in New England waters (Rough, 1983). The recolonization of New England waters by grey seals from Atlantic Canada (Sable Island and Gulf of St Lawrence stocks; Wood *et al.*, 2011) has resulted in the establishment of two major pupping colonies in the Cape Cod region (Muskeget Island and Monomoy National Wildlife Refuge) and three colonies along the mid-coast Maine. Muskeget is the largest colony with approximately 3000 pups born in the 2013/2014 pupping season. There is now a year-round population of at least 20 000 grey seals (minimum counts) in New England waters (G. Waring, pers. comm.) The increase in grey seal numbers in the region means that the species now shares or has “taken over” haul-out sites that were previously exclusively used by harbour seals (see Haul-out site usage section), although the overall range of harbour seals in coastal New England has not changed in the past century (Waring *et al.*, 2010). A 2012 survey of adult harbour seals in New England waters estimated 24% fewer compared to the previous estimate from 2001 (Waring *et al.*, in review 75 834; CV= 0.15vs.99 340; CV= 0.097), but this decline was not statistically significant. Potential reasons for the discrepancy between the two surveys include a reduction in the number of non-pups in the study area available to be counted, but also that the estimate represents real population decline (Waring *et al.*, in review). The authors note

that the potential impact of increasing grey seal population on harbour seals needs to be examined.

5.2.6 Summary

Where grey and harbour seals overlap, there is spatial variation in their population trajectories: in some regions both species are increasing (e.g. Wadden Sea, Baltic Sea and Kattegat, Ireland, France) while in other regions harbour seals are declining while grey seal numbers are on the rise (e.g. North Sea, Orkney, Sable Island). In still other areas, both species population abundances appear to be stable (e.g. west coast of Scotland). It is axiomatic that increasing numbers of grey seals in regions where the two species' terrestrial and aquatic distributions overlap will increase the likelihood of their interaction, but extrapolating cause and effect from contrasting population trends is not advised.

5.3 At sea habitat use

Harbour seals' foraging areas have been reported to be located close to the haul-out sites, often within a distance of 5–30 km (Tollit *et al.*, 1998; Vincent *et al.*, 2010; Cordes *et al.*, 2011; Cronin, 2011). In contrast, grey seals' primary foraging areas seem to be usually located further offshore, with distances ranging from 40 to over 100 km from shore (McConnell *et al.*, 1999; Sjöberg and Ball, 2000; Bowen *et al.*, 2006; Cronin *et al.*, 2012; 2013; Lidgard *et al.*, 2012; Oksanen *et al.*, 2014). Foraging behaviour of both species varies spatially however (Sharples *et al.*, 2009), and in some places foraging distributions can overlap, at least partly (Figures 5.3 and 5.4; Thompson *et al.*, 1996; Jones *et al.*, 2013; Mikkelsen *et al.*, in prep). The potential overlap in foraging areas of both species in the more inshore waters as well as partial overlap in diet (see Wilson and Hammond, 2015b) imply a potential for competition for prey resources between the two species. This hypothesis was particularly suggested in places where harbour seals have been declining while grey seal numbers were increasing (Thompson *et al.*, 2001; Bowen *et al.*, 2004; Mikkelsen *et al.*, in prep). When reviewing the possible causes for the decline of harbour seals in the UK, Hall and Kershaw (2012) highlighted that, despite some evidence of spatial overlap between the species that has been reported (Thompson *et al.*, 1996; Jones *et al.*, 2013), evidence of direct exclusion is lacking.

Svensson *et al.* (2012) modelled potential scenarios of competing harbour and grey seals in the Baltic Sea, for different competition strengths and different frequencies of Phocine Distemper Virus (PDV) epidemics. Their results suggest that the harbour seals are resilient to competition while exerting a negative effect on grey seal abundance under moderate to strong competition. Hence Baltic grey seal would benefit from weaker levels of competition. Under moderate and strong competition grey seal abundance would be a direct function of the PDV frequency as this reduces the competitive strength of harbour seals.

There are also recent reports of an influenza outbreak (Avian Influenza Virus) in northern Europe ongoing since April 2014 (Zohari *et al.*, 2014; Krog *et al.*, 2015). The outbreak appears to be affecting harbour seals only (similar to PDV) and thus might have a disproportionate impact on this species.

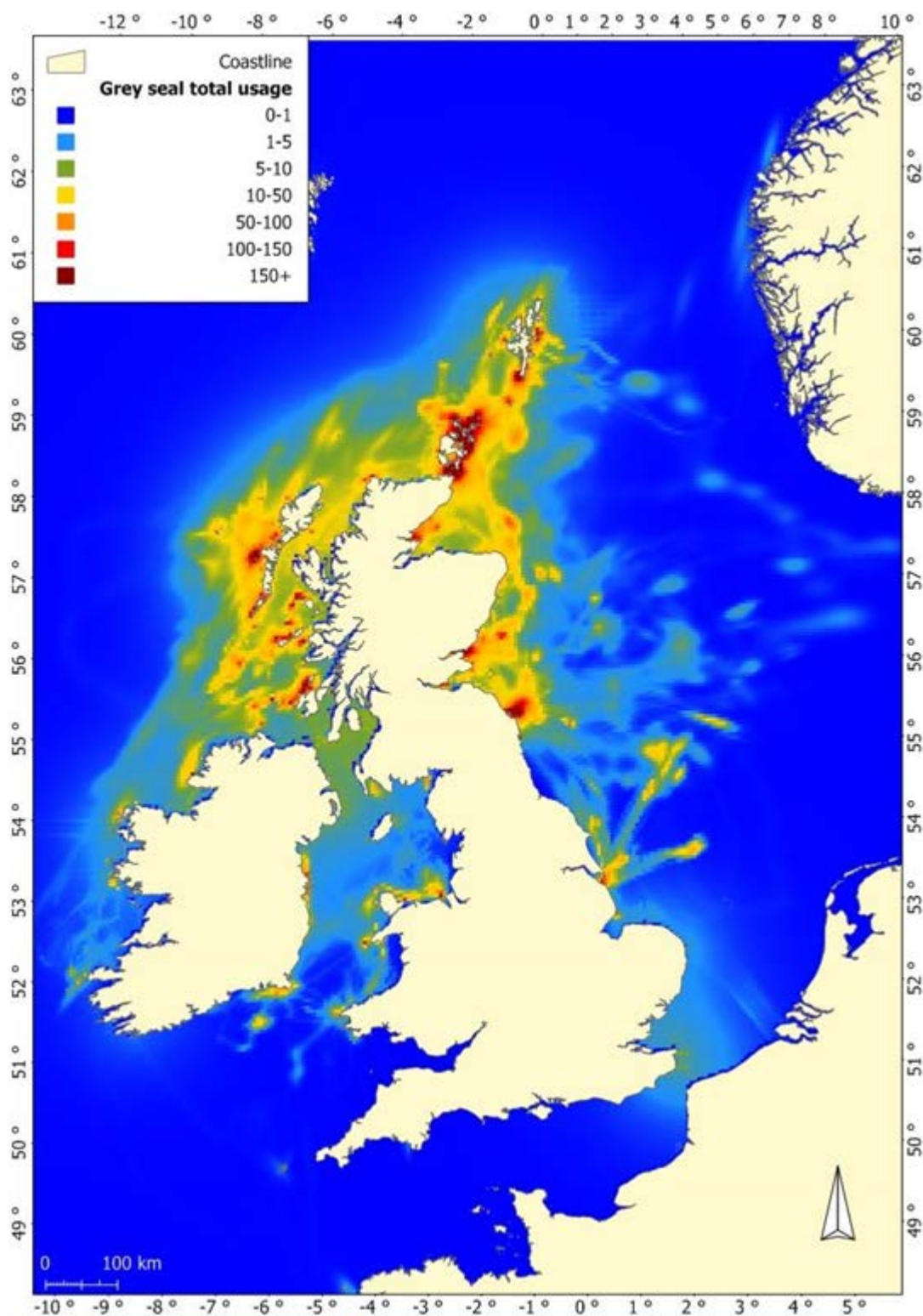


Figure 5.3. Estimated total usage (at-sea and hauled-out densities from telemetry data and aerial surveys) by grey seals around the UK (from Jones *et al.*, 2013).

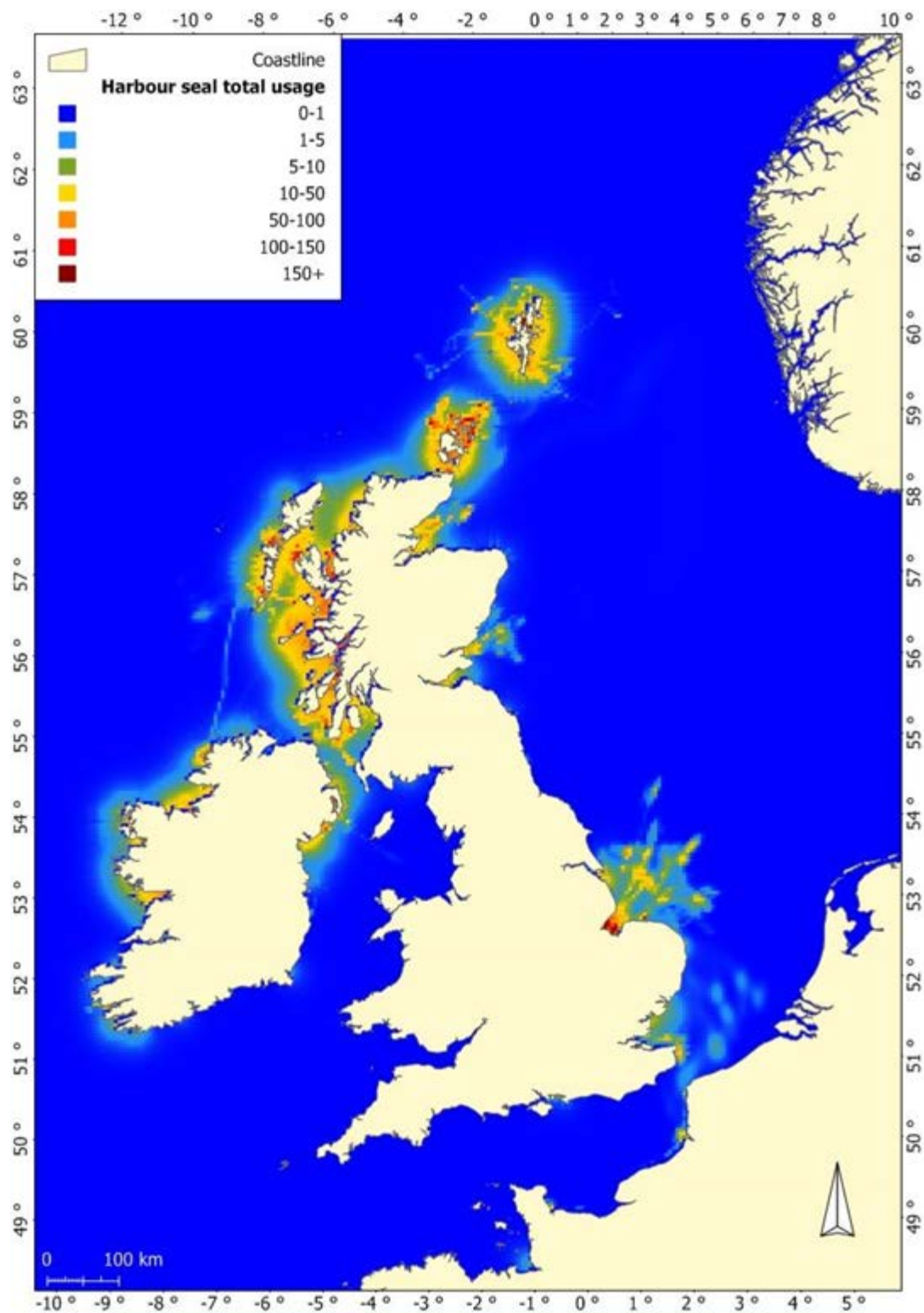


Figure 5.4. Estimated total usage (at-sea and hauled-out densities from telemetry data and aerial surveys) by harbour seals around the UK (from Jones *et al.*, 2013).

5.4 Competition for prey resources

5.4.1 Wadden Sea, Baltic Sea and Kattegat

There are no available studies to compare harbour seal and grey seal diets in the Wadden Sea. In the southwestern Baltic and Kattegat, molecular analysis of scats of both species indicated pronounced differences and modest overlaps in diet between the two species (Pittman *et al.* in prep.).

5.4.2 UK and Ireland

In a review of grey and harbour seal diet studies in the UK and Ireland, Brown *et al.* (2012) found no consistent differences in the diet of the two species but did show that there is considerable spatial and temporal variation in consumption of gadids, perciforms and flatfish. The authors highlight the need for routine scat or molecular analysis of diet as the evidence is often patchy and short term.

Much has changed in the seas around Scotland (and the rest of the UK) in the last quarter of a century. Grey seal numbers continue to rise in the North Sea while harbour seal numbers have declined in Shetland, Orkney and SE Scotland. Stocks of gadid fish have also declined. Reduced availability of a sufficient quantity or quality of prey is one potential contributory cause of harbour seal declines. Wilson and Hammond (2015b) examined the diet of both species among regions that exhibit different population trajectories. In this study, grey and harbour seal diet composition was assessed seasonally and regionally throughout Scotland and along the east coast of England over a twelve month period in 2010/2011, through quantitative analysis of hard prey remains recovered from scats.

In the North Sea, grey seal diet was dominated by sandeel in all regions. Sandeel were also dominant in the diet of harbour seals in the Moray Firth but, to the south, their diet was more varied and also included flatfish, sandy benthic and large gadid prey. In Orkney and Shetland, harbour and grey seal diet comprised mostly sandeel, large gadids and pelagic prey. The largest differences between the seal species were in Orkney in spring/summer, where harbour seals ate more sandeel, and in Orkney in autumn/winter and both seasons in Shetland where harbour seals ate more pelagic fish. West of Scotland, large gadids were the main prey in the diet of both harbour and grey seals. Pelagic fish were also important in harbour seal diet and sandeel and sandy benthic prey in grey seal diet.

Although there was considerable seasonal and regional variation in the diet of both harbour and grey seals, there was no unequivocal pattern in differences in diet between species in regions where harbour seals have and have not declined. There is thus no clear evidence that the health of harbour seal populations is influenced by diet composition.

However, there is some evidence that sandeel may play an important role in the diet of harbour seal populations in the North Sea and Northern Isles. In areas where sandeel remains important in the diet of harbour seals, their populations are at least stable but, where sandeel populations have declined, sandeel has mostly decreased in importance in the diet and harbour seal populations are also declining. The importance of gadids in the diet of harbour seals and their increased importance for grey seals in the northern North Sea and Northern Isles may be linked to the decline in importance of sandeel in the diet.

5.4.3 Western North Atlantic

While there is no direct evidence of competition for prey resources between grey and harbour seals in the Western Atlantic, there are a few pieces of indirect evidence that might suggest such an interaction. On Sable Island, Bowen *et al.* (2004) documented a long-term reduction in fecundity and female recruitment of harbour seals in conjunction with an exponential increase in the grey seal population. The authors cite interspecific competition for prey resources as a potential contributor to the harbour seal decline, in addition to mortality caused by shark attacks. Second, recent increases in grey seal bycatch in conjunction with decreased bycatch of harbour seals in the western North Atlantic might suggest that grey seals are displacing harbour seals from foraging grounds, although direct evidence of such a link is lacking (G. Waring, pers. comm.).

5.5 Haul-out site use

The two seal species are seen hauling out together in a number of areas. Sometimes the two species use distinct haul-out sites within the same area (e.g. Bowen *et al.*, 2004). Mixed haul-out groups of grey and harbour seals can also be found, but fine scale segregation is then often observed: in a number of places grey seal tend to be in the middle of the haul-out group while harbour seals tend to haul out at the periphery (Murray, 2008). In such mixed groups, behavioural interactions between the two species have been reported to be detrimental to harbour seals.

In Cape Cod (USA), Murray (2008) attributed variations in abundance of harbour seals to displacement by grey seals. She described behavioural interactions between individuals, within and between both species. Her results showed that harbour seals were more affected by disturbance from other seals. The majority of interactions were with conspecifics and harbour seals interacted at a higher frequency with other harbour seals when grey seals were present. She concluded that with increasing numbers of grey seals, increasing interspecific interactions would be inevitable and would probably displace harbour seals from prime haul-out sites. While harbour seal might continue to share haul-outs with grey seals, this may increase the frequency of more aggressive responses of harbour seals to conspecific disturbance, as well as impact position of harbour seals towards the periphery of the haul-out groups, where they are more vulnerable to conspecific aggressive behaviour.

In a similar behavioural study in Maine however, Renner (2005) found that grey seals had little, if any, effect on harbour seal behavioural patterns. He also pointed out that, although grey seals did not affect harbour seal behaviour patterns, male grey seals clearly were dominant to harbour seals during aggressive interactions. They concluded that harbour seals may recognize and avoid confrontations with the larger grey seals and that this may explain the less frequent response of harbour seals to grey seals' disturbance than to conspecific interactions.

Aggressive confrontations between the two species have also been reported in the St Lawrence estuary (Canada) and been suggested as a form of competitive exclusion (Robillard *et al.*, 2005). In Massachusetts (USA), grey seals are believed to have displaced harbour seals in at least one or possibly two haul-out sites (DiGiovanni *et al.*, 2011).

In the Eastern Atlantic, no such behavioural interactions between grey and harbour seals on land have been published. Recent observations in northern France however report grey seals disturbing harbour seals on their haul-out sites, pushing them into the water in order to haul-out there themselves. These observations were made on

small harbour seal haul-out sites and seemed to involve only male grey seals, while female grey seals did not display aggressive interactions (Karpouzopoulos and Brévar, pers. comm.). In Ireland, there is fairly distinct terrestrial niche segregation with both species co-occurring at only a few sites (M. Cronin, pers. comm.).

5.6 Predation of harbour seals by grey seals

Grey seals have been known to cannibalise grey seal pups (e.g. Bedard *et al.*, 1993; Boyle, 2011) and recently there have been reports of adult and juvenile harbour seals, and grey seal pups stranded with characteristic spiral lesions both in North America (Lucas and Natanson, 2010) and Europe (Bexton *et al.*, 2012). Until recently, the causes were hypothesized to be predation by sharks, and/or mortality resulting from collision with ducted propellers, respectively. Recent evidence from Germany and the UK shows that larger grey seals can also cause these 'corkscrew' injuries (van Neer *et al.*, 2014; Thompson *et al.*, 2015).

Van Neer *et al.* (2015) provide a detailed account of a young male grey seal predating upon young harbour seals in 2013 and 2014 near the island of Düne, in the German archipelago of Helgoland. Blubber and other tissue were missing from harbour seal carcasses; this and the fact that the grey seal was observed predating on the harbour seal in the water rather than on the beach suggest that this was feeding behaviour, rather than aberrant sexual behaviour as has previously been suggested in similar cases. The authors suggest that the rapid increase in grey seal numbers in the area is likely to be accompanied by an increase in competitive interactions between grey and harbour seals, as well as an increase in the probability of observing an uncommon behaviour.

A similar feeding behaviour was subsequently documented at the Isle of May grey seal breeding colony where an adult male grey seal was observed killing and preying upon several weaned grey seal pups in November–December 2014 (Thompson *et al.*, 2015). A total of 14 pup carcasses with characteristic spiral lesions were discovered over the research season; five of these cannibalism events were observed directly (video recorded) and attributed to the same male grey seal. The male grey seal responsible was tagged on the Isle of May and it subsequently travelled east across the North Sea to another known grey seal breeding colony in Germany, approximately 40 km from the site in Helgoland where a male grey seal was observed predating on harbour seals (van Neer *et al.*, 2015).

Post-mortems conducted on eleven of the 14 carcasses from the Isle of May showed that for nine individuals the pathology of the injuries was consistent with those described elsewhere as 'corkscrew' injuries (Bexton *et al.*, 2012). While there is no direct evidence of grey seals predating on adult harbour seals as yet, it seems likely that the behaviour is responsible for a proportion of the harbour seal deaths due to 'corkscrew' injuries reported in recent years (Thompson *et al.*, 2013).

Predation on harbour seals by grey seals is thus an example of a direct ecological interaction between the two species, although its prevalence and impact on harbour seal populations is uncertain. While it is an underestimate of the true mortality, the reported number of individual harbour seals with characteristic wound patterns is small (<100 in the UK; Thompson *et al.*, 2013). It is unlikely that the specialisation of a few grey seals on harbour seals would have large-scale population consequences for harbour seals. However, any reduction in adult and juvenile harbour seal survival in small, declining populations could have a significant negative impact on their recovery (e.g. Firth of Tay) and this behaviour may also contribute to a reduction of har-

bour seal use of haul outs shared with grey seals for resting and particularly pupping.

Similarly to predation of grey seals on harbour seals, recently it has been suggested that grey seals prey on harbour porpoises (Haelters *et al.*, 2012), a theory quickly followed by evidence from observations (Bouveroux *et al.*, 2014) and finds of grey seal DNA in the lesions of mutilated harbour porpoises washed ashore (van Bleijswijk *et al.*, 2014; Jauniaux *et al.*, 2014). The temporal and spatial scale of the phenomenon, together with the numbers of harbour porpoises washing ashore with indication of death due to grey seal interaction (Leopold *et al.*, 2014; 2015; Stringell *et al.*, in press), indicate that the predation cannot be limited to a small number of grey seal bulls having partly switched their diet to harbour porpoises. The phenomenon, for which anecdotal evidence suggests that it has occurred on a small scale before the first cases were described, concurs, at least in the southern North Sea, with local increases in both harbour porpoises and grey seals. The lesions observed in the harbour porpoises supposed or proven to have been killed by grey seals (Haelters *et al.*, 2012; Bouveroux *et al.*, 2014; Leopold *et al.*, 2014), are in many cases very similar to the lesions observed in harbour seals killed by grey seals (van Neer, 2015; Thompson *et al.*, 2015). A different cause of death has been attributed to similarly mutilated seals of different species in Canada (a.o. Sable Island), and the WGMME advises that these cases should be re-evaluated in the light of emerging evidence of grey seal predation.

Recommendation: Given the emerging strong evidence of grey seal predation on harbour seals, which potentially affects populations across the ICES region, the WGMME recommends that a dedicated workshop as a mechanism with which to develop a coordinated research plan for North Atlantic seals.

6 ToR d: Update the database for seals, and review its usefulness in meeting the needs of Member States under EU's MSFD

6.1 Introduction

There is a standing requirement to collate information on seal population monitoring across the ICES area into the Seals Database, and to keep the new seal database up to date and to ensure its usefulness. Furthermore, the database will form the primary source of information with which to assess seal population abundance and distribution against targets set under the Marine Strategy Framework Directive (MSFD).

6.2 Area of relevance

The area of relevance to ICES is the North Atlantic, including the North and Baltic Seas.

To date, the ICES seal database includes census information from a few countries in the Greater North Sea, and to a lesser extent to the Celtic Seas. A separate seal database is being compiled and further developed by HELCOM for the Baltic area, and another database was recently formalised in the USA through the Atlantic Marine Assessment Program for Protected Species (AMAPPS).

In order to fulfil the MSFD's requirements, it is suggested that the existing ICES seal database be completed by all member states within the Celtic Seas and Greater North Seas, i.e. the UK, Ireland, Denmark, Norway, Sweden, Germany, the Netherlands, Belgium and France. As the Greater North Sea region includes the Limfjord and Kattegat regions, which are already covered by the HELCOM for harbour seals, it is acknowledged that the data for this species might be included in both databases. Grey seal data in these areas might only be included in the ICES database, based on genetic evidence and movement patterns observed across the North Sea. All other seal data from the Baltic Sea would therefore be included in the HELCOM database.

6.3 Issues

The information currently contained in the database is insufficient. Most issues regarding the structure of the database and what data needs to be submitted to the database were thoroughly discussed in the WGMME 2014 (see Section 7). This year, we re-iterate the need for member states to populate the database with relevant information by the end of 2015 to allow for preliminary assessment in 2016. A dedicated workshop was held March 18th/19th 2015 at the University of St Andrews, which aimed to (1) establish distribution metrics and targets and baselines for both abundance and distribution and (2) identify the data required to produce an Intermediate Assessment in 2017.

Use of seal survey data for assessment of MSFD distribution targets

At present, no information with which to assess changes in seal distribution is reported to the ICES seal database, although the MSFD indicator M-3 'Abundance and distribution of harbour and grey seals'¹ requires such information. Any semi-valid

¹ This indicator is the result of a merger of two common indicators: M1: seal distribution and M3: seal abundance. The two indicators were merged based on recommen-

assessment of changes in species distribution or distributional pattern requires both spatially explicit reporting of animal abundances (i.e. coordinates of haul out locations) and an estimate or measure of sampling effort. Still, the WGMME stresses that robust assessments of changes in pinniped distribution from surveys that are designed primarily to estimate abundance are problematic.

As the seal surveys are designed for assessing abundance, the data produced carry some inherent limitations regarding their use for assessment of distribution. First, the surveys only cover distribution on land (except for the ringed seal, which is surveyed on sea ice) and do not address the distribution at sea. Second, the surveys do not cover potential haul-out sites or breeding colonies in a systematic way. Haul outs and breeding sites are sampled preferentially based on past experience of seal occurrence. This means that the surveys will not detect expansions of ranges; new haul outs are only added to the survey coverage as anecdotal data on seal occurrence accumulate. Third, the surveys only cover narrow windows during key moments of the life cycle of the seals, namely peak moulting and pupping seasons.

Survey data may be used to detect contractions in range in terms of reduced use or abandonment of haul outs or areas, depending on the resolution with which data are reported. Temporal shifts in distribution density within the area covered by the surveys can be described at the spatial resolution provided in the data. This spatial resolution needs to be at least at the level of the Assessment Units and preferably at smaller scales for both species.

Recommendation: The WGMME notes the need for the relevant authorities from the OSPAR Maritime Area provide data to populate the seal database at a time requested by the OSPAR Commission, so that they can be included in an assessment of seal Common Indicators for OSPAR's Intermediate Assessment 2017.

dations of ICG-COBAM to BDC 2014: *"There is no need to have M1 as common, as the results (distribution pattern of seals) will follow from the results of the indicator M3 (abundance of seals)."* (See Annex 2.1 in BDC Paper 14/2/2-E.).

7 ToR e: Review and evaluate multispecies models that incorporate marine mammal consumption to assess marine mammal impacts on fishery resources, and make recommendations for improvements in input data and assumptions for the North Atlantic

7.1 Introduction

Marine mammals have been included within a wide range of multispecies models (Table 7.1) and the approaches used vary substantially in terms of both scope and objectives. Minimum realistic models and extended single species assessment models are, as the names suggest, focused on answering particular questions about the role of specific species and their parameters are estimated statistically. Modelling approaches such as Atlantis or Ecopath with Ecosim (EwE) on the other hand, often aim to represent a large proportion of ecosystem components and in these circumstances it becomes harder to estimate model parameters through comparisons with real data. As such, EwE models routinely use parameter values derived for other ecosystems, especially for lower trophic levels.

Whether and to what extent marine mammals compete with fisheries has long been a contentious political and scientific issue. Demonstrating that competition exists is difficult (much research, e.g. on marine mammals diets, provides only circumstantial evidence), and quantifying such interactions, and identifying management measures that successfully balance conservation and fishery objectives, is challenging. Model-based approaches offer the potential to achieve all these objectives without the need for large-scale and potentially damaging manipulations of the ecosystem. However, their validity depends on the robustness of the assumptions underlying the models and marine ecosystem models are notoriously difficult to construct because of their complexity as well as the many unknowns concerning fundamental ecosystem processes.

Aside from issues of data availability and parameter estimation, which are more acute for more complex models, a particular concern relates to the assumptions made about predator functional responses. In the real world, most predators would be expected to adjust their diet following changes in abundance of their prey (especially the preferred prey). Functional responses, especially multivariate functional responses, are difficult to estimate from field data but even a simplistic assumption, that relative importance of each prey species in the diet follows its relative availability to the predator, would be more realistic than assuming a fixed diet. Assumptions about the forms of predator functional responses, both for individual predators and the population of predators (in the latter case incorporating the “numerical response” and potentially considering intraspecific competition) are expected to have significant effects on system behaviour (Plagányi and Butterworth, 2004; Smout *et al.*, 2010; Hunsicker *et al.*, 2011). Christensen and Walters (2004) argue that EwE can adequately represent predator functional responses but if a single set of diet data is assumed to apply across a range of prey abundance conditions, prey selection will be misrepresented (e.g. Plagányi and Butterworth, 2004).

Plagányi (2007) assessed a number of models in terms of various requirements, including the requirement that marine mammals be explicitly included, rather than treated as exogenous components. She concluded that Globally Applicable Area Disaggregated General Ecosystem Toolbox (GADGET) and Minimum Realistic Models (MRM), such as the approach of Punt and Butterworth (1995), showed the most

promise as tools to assess indirect interactions between marine mammals and fisheries. Bioenergetic modelling approaches such as that of Koen-Alonso and Yodzis (2005) also have a role to play to characterize the finer details of these interactions. In the context of marine mammal-fisheries interactions, NAMMCO (2003) considered that GADGET has potential utility for management, for testing scenarios in which abundances of target species are manipulated and to identify the inputs which had the greatest effect on model predictions and hence to guide research priorities. A systematic evaluation of different modelling approaches for the Icelandic continental shelf area (and other areas) is being undertaken within the MareFrame project.

Table 7.1. The range of multispecies models in which marine mammals have been included.

CATEGORY	EXAMPLES
1. Whole ecosystem and dynamic system models (WEM)	1.1. Ecopath with Ecosim (EwE) 1.2. ATLANTIS 1.3. EcoNetwrk 1.4. MARXAN 1.5. Other WEM
2. Minimum realistic models (MRM)	2.1. ESAM 2.2. MSVPA-MSFOR and MSVPA-X 2.3. MULTSPEC, BORMICON and GADGET 2.4. SMS 2.5. Size-spectrum models 2.6. Multispecies Production Models 2.7. Other MRM
3. Individual-based models (IBM)	3.1. OSMOSE 3.2. Other IBM
4. Bioenergetic models (BM)	4.1. BM

The present review is based mainly on models constructed for the ICES area and is organized by model type and region.

7.2 Whole ecosystem and dynamic system models

7.2.1 Ecopath with Ecosim (EwE)

Ecosim is a biomass-dynamic simulation routine for Ecopath ecosystem models (Walters *et al.*, 1997; Pauly *et al.*, 2000; Christensen and Walters, 2000) and has been proposed as a useful tool for exploring interactions between cetaceans, their prey, and fisheries. However, although applications exist (e.g. Lassalle *et al.*, 2012), its usefulness for addressing such questions remains to be fully tested.

As mentioned above, the way that Ecosim represents how predators respond to changes in prey availability (functional response) is an issue of particular importance. There is evidence from several fisheries and plankton models that the type of functional response specified can greatly affect model predictions (Magnusson and Pals-son, 1991; Steele and Henderson, 1994; Mohn and Bowen, 1996; Tett and Wilson, 2000).

Uses of EwE models include exploring interactions between marine mammals, their prey and fisheries (e.g. Blanchard *et al.*, 2002; MacKenzie *et al.*, 2011; Tomczak *et al.*, 2011; 2012; Lassalle *et al.*, 2011), investigating the relative importance of bottom-up and top-down effects (e.g. Skaret and Pitcher, 2007), comparing trophic structure

across ecosystems (e.g. Gaichas *et al.*, 2009), and predicting community responses to natural or anthropogenic environmental change (e.g. Niiranen *et al.*, 2008), management actions (e.g. Hansson *et al.*, 2007; Gårdmark *et al.*, 2013) or combinations of environmental changes and management actions (Österblom *et al.*, 2007). EwE models have also been used to explore effects pollutant bioaccumulation on humans (Booth and Zeller, 2005).

One of the most comprehensive models is that developed for the North Sea by Mackinson and Daskalov (2007) which includes seals and various cetacean species and has been employed in numerous subsequent studies to address topics such as (a) the relative importance of climate and fishing in determining ecosystem structure and primary productivity (Mackinson *et al.*, 2009a), (b) evaluate Maximum Sustainable Yield (MSY) within a multispecies context (Mackinson *et al.*, 2009b), (c) the size, spacing and impacts of MPAs (Le Quesne *et al.*, 2008), (d) trade-offs in management scenarios, (Mackinson *et al.*, 2009b), (e) development of ecosystem indicators of change (e.g. Lees and Mackinson, 2007), (f) investigating the rebuilding of fish stocks (Worm *et al.*, 2009), (g) the effects of subsidies on North Sea fisheries (Heymans *et al.*, 2011), (h) the role of forage fish in marine ecosystems (Smith *et al.*, 2011) and (i) prediction of past trends in abundance of all ecosystem components (Mackinson, 2013).

Although widely used, Ecopath and EwE have also been criticised by many authors, partly for the use of literature values where empirical values are not known, partly for unrealistic assumptions. However, at least in principle, outputs can be compared to empirical data, e.g. in relation to abundance trends or effects of management actions, affording opportunities to validate the models. In addition, as several authors have done, EwE models can be compared with other ecosystem models.

Table 7.2. Ecopath and EwE models including marine mammals.

REGION	MARINE MAMMAL GROUPS	REFERENCES
Barents Sea	Minke whale, other baleen whales, toothed whales, seals	Blanchard <i>et al.</i> (2002)
Barents Sea	Grey seal	MacKenzie <i>et al.</i> (2011)
Norwegian Sea and Barents Sea	Marine mammals	Dommasnes <i>et al.</i> (2001)
Norwegian Sea and Barents Sea	Marine mammals	Skaret and Pitcher (2007), Gaichas <i>et al.</i> (2009)
Sørkjørd, Norway	Harbour porpoise, seals (two species), sea otter	Pedersen <i>et al.</i> (2008), Morissette <i>et al.</i> (2009a)
Baltic Sea	Ringed seal, grey seal	Harvey <i>et al.</i> (2003), Hansson <i>et al.</i> (2007), Österblom <i>et al.</i> (2007)
Baltic Sea	Seals	Sandberg, 2007
Baltic Sea	Seals	Niiranen <i>et al.</i> (2008)
Central Baltic Sea	Seals	Tomczak <i>et al.</i> (2009a,b, 2011, 2012, 2013); Gårdmark <i>et al.</i> (2013); Lassalle <i>et al.</i> (2013)
Baltic Sea	Seals	Niiranen <i>et al.</i> (2012, 2013); Tomczak <i>et al.</i> (2012)
Iceland	Baleen whales, toothed whales, seals	Mendy and Buchary (2001); Buchary (2001)
Faroe Islands	Two marine mammal groups	Zeller and Freire (2001); Zeller and Reinert (2004); Booth and Zeller (2005)
North Sea	Seals	Christensen <i>et al.</i> (2002); Beattie <i>et al.</i> (2002); (Morissette <i>et al.</i> , 2012)
North Sea	Seals, various cetaceans	Mackinson (2002a, 2002b)
North Sea	Seals, various cetaceans	Mackinson and Daskalov (2007); Lees and Mackinson (2007); Le Quesne <i>et al.</i> (2008); Mackinson <i>et al.</i> (2009a b); Worm <i>et al.</i> (2009); Heymans <i>et al.</i> (2011); Smith <i>et al.</i> (2011); Mackinson (2013); Lassalle <i>et al.</i> (2013)
Southern North Sea	Marine mammals	Staebler <i>et al.</i> (2014)
West coast of Scotland	Harbour seal, grey seal, cetaceans	Haggan and Pitcher (2005); Bailey <i>et al.</i> (2011); Alexander <i>et al.</i> (2015)

REGION	MARINE MAMMAL GROUPS	REFERENCES
Irish Sea	Marine mammals	Lees and Mackinson (2007)
Western English Channel	Toothed whales, seals	Stanford and Pitcher (2004); Araújo <i>et al.</i> (2005, 2006)
Eastern English Channel	Various cetaceans	Villanueva <i>et al.</i> (2009); Georgi <i>et al.</i> (2011)
Celtic Sea and Bay of Biscay	Baleen whales, toothed whales	Guénette and Gascuel (2009a,b)
Celtic Sea	Marine mammals	Lauria <i>et al.</i> (2010); Lauria (2012)
Bay of Biscay	Marine mammals	Ainsworth <i>et al.</i> (2001)
Central Bay of Biscay	Five odontocetes	Lassalle <i>et al.</i> (2011, 2012, 2013, 2014a,b,c)
Gulf of Cadiz	Killer whale, dolphins	Torres <i>et al.</i> (2013)
Azores	Baleen whales, false killer whale, other odontocetes	Guénette and Morato (2001); Morato <i>et al.</i> (2014)
West Greenland	Baleen whales, toothed whales, seals	Pedersen and Zeller (2001)
Lancaster Sound (Canada)	Polar bears, killer whale, narwhal, beluga, bowhead whale, harp seal, bearded seal, ringed seal, walrus	Mohammed (2001)
Labrador Shelf, Northeast Newfoundland Shelf and Grand Bank	Hooded seal, harp seal	Bundy <i>et al.</i> (2000); Bundy (2001)
Eastern Scotian Shelf	Cetaceans, seals	Bundy and Guénette (2002); Bundy (2004, 2005)
Northern Gulf of St Lawrence	Harp seal, grey seal, hooded seal	Morisette <i>et al.</i> (2003, 2006, 2012); Savenkoff <i>et al.</i> (2004a)
Northern Gulf of St Lawrence	Seals, cetaceans	Savenkoff <i>et al.</i> (2009)
Southern Gulf of St Lawrence	Harp seal, grey seal, hooded seal	Savenkoff <i>et al.</i> (2004b, 2007)
Western Scotian Shelf, inner waters of the Gulf of St Lawrence	whales, toothed cetaceans, seals	Araújo and Bundy (2011)
Middle Atlantic Bight (USA)	Dolphins, porpoise, baleen whales, seals	Okey (2001)
Cape Hatteras, North Caro-	Baleen whales	Okey and Pugliese (2001)

REGION	MARINE GROUPS	MAMMAL	REFERENCES
lina to Florida Keys			
Gulf of Maine, Georges Bank, Southern New England and Middle Atlantic Bight	Baleen whales, odontocetes, pinnipeds	odon-	Link <i>et al.</i> (2006, 2008); Overholtz and Link (2009)
NE US continental shelf	Baleen whales, odontocetes, pinnipeds	odon-	Gaichas <i>et al.</i> (2009); Megrey <i>et al.</i> (2009); Link (2010)
Oceanic waters of the North Atlantic	Baleen whales, beaked whales, other odontocetes		Vasconcellos and Watson (2004)

7.2.2 Atlantis

Atlantis is a modular ecosystem model, developed by the *Commonwealth Scientific and Industrial Research Organisation (CSIRO)* of Australia, which considers all parts of marine ecosystems, biophysical, economic and social. Thus, components include solar radiation, hydrodynamics, growth, migration, fishery capture, fleet dynamics, market values and regulation (Fulton *et al.*, 2004). The modelling environment includes a virtual ocean with all its complex dynamics, a virtual monitoring and assessment process, a virtual set of ocean-uses (namely fishers), and a virtual management process. In addition Atlantis models can include both horizontal and depth stratification. Although parameterized, initialized and loosely tuned to empirical values, Atlantis is very complex and was not designed to provide specific tactical management advice for a particular stock (e.g. a quota or effort limit). Rather, it is a research tool and a simulator to guide strategic management decisions and broader concerns.

Atlantis has been used to explore consequences of removing species, and to provide multispecies fishery advice and multi-sector ocean-use advice. Examples are listed in Table 7.3. The chief drawback of Atlantis is that it is unwieldy in its complexity and consequently takes a long time to parameterize, initialize, calibrate, and run any particular application. Additionally, the validation routines available for Atlantis are currently very limited.

Table 7.3. Atlantis models that include marine mammals.

REGION	MARINE GROUPS	MAMMAL	REFERENCES
Barents Nordic and Baltic Seas	Main marine groups	mammal	Nielsen <i>et al.</i> (2014), Palacz <i>et al.</i> (2014, 2015)
North Sea and English Channel	Baleen whales, toothed whales, seals		(see ICES, 2011b, 2013c, 2014c)
Northeastern States	United Pinnipeds, whales, toothed whales	baleen whales	Link <i>et al.</i> (2011)

7.2.3 Econetwrk

Network analysis is a phenomenological approach that quantifies the structure and function of foodwebs by evaluating biomasses and energy flows. In principle, this includes four types of question (see <http://www.glerl.noaa.gov/econetwrk/>: (1) input/output analysis, (2) determination of trophic status and identification of food chains, (3) elaboration and analysis of biogeochemical cycling and the supporting flows, and (4) calculation of ecosystem indices (or properties), derived from information theory, that describe the state of the foodweb.

The efficiency with which energy and material is transferred, assimilated, and dissipated conveys significant information about the structure and function of foodwebs (Ulanowicz and Platt, 1985; Baird and Ulanowicz, 1989; 1993; Baird *et al.*, 1991; Ulanowicz and Wulff, 1991; Ulanowicz, 2004). Network analysis has been used to compare ecosystems of different size, geographical location, hydrological characteristics, and trophic status (Baird *et al.*, 1991; Ulanowicz and Wulff, 1991; Baird and Ulanowicz, 1993; Monaco and Ulanowicz, 1997). Most recently, arguments have been advanced for the use of network analysis to quantify the health and integrity of ecosystems (Ulanowicz, 2000) and to evaluate the magnitude of stress imposed on an ecosystem (Ulanowicz, 1995; Mageau *et al.*, 1998).

The algorithms used by EcoNetwrk and Ecopath are based on the same concepts, albeit with some differences in interpretation. There are however fundamental differences in data input, which lead to differences in the output. For example, EcoNetwrk takes the respiration of primary producers into consideration, while Ecopath does not. This leads to various discrepancies in the calculation of throughput and all the parameters related to it, such as the 'ascendancy' and 'development capacity'. In most cases, the differences are small enough that the same qualitative conclusion would result from using either Ecopath or EcoNetwrk. In practice they can be viewed as complementary approaches with different strengths and weaknesses. Heymans and Baird (2000) and Link *et al.* (2006) compared the outputs of Ecopath and Econetwrk models for the northern Benguela ecosystem and Georges Bank (USA) respectively.

7.2.4 MARXAN

MARXAN was designed as a conservation planning tool, on the existing SPatially EXplicit Annealing (SPEXAN) program (Ball *et al.*, 2009). It uses stochastic optimisation routines (Simulated Annealing) to generate spatial reserve systems that achieve particular biodiversity goals with reasonable optimality. Computationally, MARXAN provides solutions to a conservation version of the knapsack problem (optimal resource allocation under financial constraints), where the objects of interest are potential reserve sites with given biological attributes. The simulated annealing algorithm attempts to minimise the total cost of the reserve system, while achieving a set of conservation goals. An example for the English Channel, using a model which included marine mammals, is given by Metcalfe *et al.* (2015) who explored the consequences of different MPA management strategies in the English Channel.

7.2.5 Other WEM

Other whole ecosystem models include those based on nutrient pools, e.g. the European Regional Seas Ecosystem Model (ERSEM), as well as bioenergetic models more similar in concept to those previously described. The models listed (Table 7.4) do not currently include marine mammals but could be adapted to do so.

Table 7.4. Other whole ecosystem models.

MODEL	APPLICATION	REFERENCES
European Regional Seas Ecosystem Model (ERSEM)	Bottom-up processes impacting fish populations	Baretta <i>et al.</i> (1996); Baretta-Bekker and Baretta (1997); Heath <i>et al.</i> (1997); Lenhart <i>et al.</i> (1997)
Shallow Sea Ecological Model (SSEM)	Impact on fisheries of coastal development activities	Sekine <i>et al.</i> (1991)
Spatial Environmental POPulation Dynamics Model (SEAPODYM)	Tuna fisheries	Bertignac <i>et al.</i> (1998); Lehodey, (2001; 2005); Lehodey <i>et al.</i> (2003)
Apex Predators ECOSystem Model (APECSIM)	Basin-wide spatial dynamics of open ocean pelagic ecosystems	Maury <i>et al.</i> (2007a, b)
ECOGOMAG	Research on Gulf of Maine (GOM) ecosystem	(see ICES, 2007)

7.3 Minimum Realistic Models (MRM)

Minimum Realistic Models, as their name indicates, are models constructed to represent a limited number of system components and their interactions, under the assumption that the chosen components are the ones likely to have the most important interactions with a target species of interest (Plagányi, 2007). Typically, MRMs incorporate predator abundance, diet (stomach contents data) and consumption estimates and have been used to examine predator and fishery impacts on commercially important fish species.

MRMs have been criticized for their simplicity, with critics noting that, by focusing on a few species, model users may be missing complex interactions and non-linear responses, obtaining unrealistic results and emphasizing the potential for competition between predators and fleets without considering likely trade-off interactions. However, compared to whole ecosystem model, MRMs are evidently both easier and quicker to use, in terms of statistical fitting of the models to data, and they offer lower uncertainties around parameter estimates. Some authors consider MRMs to be the most suitable option for studying predator–prey interactions as well as for the assessment of interactions with marine mammals (Plagányi, 2007).

7.3.1 Extended Single-species Assessment Models (ESAM)

The assessment of the Norwegian spring-spawning herring stock is carried out using the SeaStar (Stock Estimation with Adjustable Survey observation model and Tag Return data) model. SeaStar is based upon traditional assessment methods and includes predation on herring by minke whales (Tjelmeland and Lindstrøm, 2005). Initial attempts to include three species of cetaceans, minke, fin and humpback whales, in a multispecies model indicate that their effect on cod and capelin stocks may be considerable (Stefánsson *et al.*, 1997).

7.3.2 MSVPA–MSFOR and MSVPA–X

Multispecies virtual population analysis (MSVPA) and multispecies forecasting (MSFOR) are multispecies stock assessment models that use catch-at-age data and fish stomach content data to estimate both past fishing mortalities and predation

mortalities (Helgason and Gislason, 1979; Sparre, 1991). The 4M package (Multi-species, Multifleet, Multi-area Model-package) (Vinther *et al.*, 2002) contains MSVPA/MSFOR and a tuning module (Vinther, 2001) and has been used to run MSVPA and MSFOR at the ICES multispecies study groups in recent years.

Examples of use of MSVPA which include predation by marine mammals include models for Iceland (with cod, capelin, Northern shrimp and whales; Steffansson *et al.*, 1997) and the North Sea (with 15 fish species, seabirds and grey seals; ICES, 2005b).

MSVPA-X or 'extended' multispecies virtual population analysis is an expanded version of MSVPA, in effect a series of single-species VPAs linked together via a feeding model (Garrison *et al.*, 2010). It is used to examine stock dynamics of multiple species that feed on each other, in particular to explore the relative importance of removals by predators and fisheries. Marine mammals are currently not included.

7.3.3 MULTSPEC, BORMICON and GADGET

GADGET (www.hafro.is/gadget) is based on an age-length structured forward-simulation model and was designed to accommodate multiple areas and fleets, mixed fisheries and predation. The inner workings of GADGET and its antecedents (MULTSPEC, BORMICON, Fleksibest) have been described by several authors, notably Bogstad *et al.* (1997), Stefánsson and Pálsson (1997), Begley (2004) and Begley and Howell (2004). MULTSPEC was developed to examine marine mammal fisheries interactions the Barents Sea (Bogstad *et al.*, 1997; Tjelmeland and Bogstad, 1998). Fleksibest is an age-length structured population model. The BOREal Migration and CONsumption model (BORMICON) was developed in the 1990s as a generalisation of MULTSPEC (Stefánsson and Pálsson, 1997) and evolved into GADGET (Begley and Howell, 2004). Howell and Bogstad (2010) linked this model to the Fisheries Library in R (FLR package) in order to perform management strategy evaluations. Examples of applications are summarized in Table 7.5.

As with all ecosystem models which aim to capture trophic interactions, availability of appropriate diet data is a key issue. Where this concerns marine mammals this has resulted in some controversy, with some nations (e.g. Iceland) collecting such data using lethal sampling while others regard this as unacceptable for protected species. Work to improve GADGET models in Iceland has also included use of tagging data to estimate migration rates (Hannesson *et al.*, 2004; 2008) and the introduction of a bootstrap technique to evaluate uncertainty (Elvarsson *et al.*, 2014).

Table 7.5. Applications of GADGET and related models which include marine mammals.

REGION	APPLICATION	MARINE MAMMALS INCLUDED	REFERENCES
Iceland	Stock assessment	Minke whale	Taylor <i>et al.</i> (2004, 2007); Taylor and Stefansson (2004); Taylor (2011); Elvarsson <i>et al.</i> (2013); Elvarsson (2013)
Barents Sea and Norway	Impact of marine mammals on fish stocks	Minke whale, harp seal	Bogstad <i>et al.</i> (1997); Tjelmeland and Bogstad (1998) Tjelmeland (2005); Tjelmeland and Lindstrøm (2005); Howell and Bogstad (2007); Lindstrøm <i>et al.</i>

			(2009)
Bay of Biscay and Iberian Peninsula	Hake assessment, anchovy and hake trophic relationships	(Inclusion of dolphins is planned)	Cerviño <i>et al.</i> (2008, 2009); Fernandes <i>et al.</i> (2010); ICES (2010c); Andonegi <i>et al.</i> (2011); Saavedra <i>et al.</i> (2013; 2014b; 2014c)

7.3.4 Stochastic Multispecies model

The Stochastic Multispecies model (SMS) (Lewy and Vinther, 2004) is a stock assessment model that includes trophic interactions, based on fitting a size-dependent food selection function. The approach contains submodels for stock–recruitment, food selection, predation mortality, fishing mortality and survey catchabilities. It is a stochastic model in which the uncertainties in fishery, survey and stomach contents data are considered.

Applications have been developed for the Baltic Sea, looking at prey seal predation on clupeid stocks (Lewy and Vinther, 2004), and the North Sea, again including grey seal but focused on predictions of climate change effects on yield and distribution (see ICES, 2012).

7.3.5 Size-spectrum models

In contrast to the species-based approach by, say, Ecopath, a size-based model assumes that the size rather than the species most heavily influences the role of an individual in an ecosystem. The approach relies on the principle that the frequency distribution of body mass follows regular patterns which can be altered by fishing, climate, etc. Size-based models are generally simpler and less data-demanding than conventional multispecies and ecosystem models. They can provide a useful tool for answering questions about effects of fishing on the structure of ecosystems. Such models have been developed for the North Sea (see Benoît and Rochet, 2004; Blanchard *et al.*, 2009; ICES 2007).

7.3.6 Multispecies production models (MS-PROD)

MSPROD is a multispecies extension of the Schaeffer production model, originally developed for Georges Bank, to include predation and competition effects. This model is parameterized with empirically based values and can be used to run scenarios to evaluate the relative importance of predation, intra-guild competition, between guild competition, and fisheries removals. It requires initial biomass estimates, carrying capacities, predation and competition interaction terms, growth rates, and fishery removals (Gamble and Link, 2009). This model has not still been used with marine mammals.

Kraken (Multispecies/Aggregate Surplus Modelling Package) is a surplus production modelling package in development for the US Northwest Atlantic (see ICES, 2013c).

7.3.7 Other MRM

Two models examine harp seal predation on fish, in the Barents Sea (Bifrost model; Tjelmeland, 2005) and Gulf of St Lawrence (SIMCAB model; Bousquet *et al.*, 2014). Pope and Holmes (2008) presented two approaches to modelling grey seal feeding on cod on the west coast of Scotland (ICES Division VIa). The first was a length-based delay difference method called proto-moments modelling. The second method in-

volved adding seal consumption-at-age estimates to the TSA model used for cod assessment. Outside the ICES area, an MRM was used to study the effect of culling Cape fur seal (*Arctocephalus pusillus pusillus*) on Benguela hake fisheries (Punt and Butterworth, 1995).

The STOCOBAR (STOCK of COd in the BAREnts Sea) model describes stock dynamics of cod in the Barents Sea, taking into account trophic interactions and environmental influences (Filin, 2005, 2007, 2010). The CODCAB model (COD and CApelin in the Barents Sea) and EFIBAR (Ecosystem and FISheries in the BAREnts Sea) models are both extensions of STOCOBAR. Marine mammals have not yet been included in this model.

The Dynamic Stochastic Foodweb (DSF) model was developed for the Barents Sea ecosystem, aiming to take a simple approach while being capable of mimicking a wide range of features observed in this ecosystem. It considers mass balance, physiology, life-history traits and stochasticity, and it includes minke whale. Despite its simplicity, the model generated realistic results (see ICES, 2011b).

tGAM is a statistical model linking plankton, fish stocks and drivers of change such as temperature and fishing pressure. Models have been developed for the North Sea, Black Sea and Baltic Sea (Llope *et al.*, 2011; Blenckner *et al.*, 2015). It is expected that the model will be used to evaluate the responsiveness of indicators (e.g. OSPAR pelagic habitat 'lifeform' indicators, trophic level of the fish community and landings and seabird productivity) to key foodweb links, climate change (sea-surface temperature) and fishing pressure but does not currently include marine mammals.

A Multispecies Integrated Stochastic Operative Model (MSI-SOM) was presented in 2013 for the Baltic Sea. The model consists of three stochastic operative models (SOMs) for cod, sprat and herring stocks, respectively. Marine mammals have not been included in this multispecies model (see ICES, 2013c).

The Hydra model (Multispecies Length-Based Operating Model) is currently under development for the US Northwest Atlantic. It is a spatial multispecies and length-structured model that includes reproductive biology, environmental covariates on growth, maturity, and fecundity, harvest, and predation (see ICES, 2013c).

7.4 Individual-based models (IBM)

Traditionally two main types of ecological models have been used: aggregate state models (like EwE) and individual-based models (IBMs) (see, for example, DeAngelis and Gross, 1992; Van Winkle, Rose and Chambers, 1993; Grimm, 1999). Formal separation of these model types is not always easy (Plagányi, 2007). For instance, within the latter form of model, the individuals may represent schools, patches of homogeneous ground cover, flocks, patches of reef, or some other subset of a population that could be treated as equivalent to an entity. Thus most aggregate state models could be seen as special cases of individual (agent) state models.

IBMs follow the fate of individuals through their life cycle, assigning to them specific features and behavioural rules, under the assumption that individual behaviour influences population dynamics. These models are sometimes referred to as "agent-based" models with the "individual/agent" being represented by either individual animals, or composite units such as fish schools or fishing fleets. The 'individual-based' approach is receiving increasing attention among ecologists (Grimm, 1999). IBMs have typically been applied to investigate the dynamics of a single population within the marine environment, but a number of applications extend these analyses

to consider multispecies dynamics. Relevant associated developments include a visualisation tool for outputs from IBM simulations (Megrey *et al.*, 2002) and a standard protocol for describing individual-based and agent-based models (Grimm *et al.*, 2006).

Advances in the use of hybrid models have seen the incorporation of a wide variety of ecosystem components into ABMs, facilitated by the coupling of classical dynamic models, using differential equations and decision-based agents. In this way, the best means of representing each ecosystem component can be used.

7.4.1 OSMOSE

OSMOSE (Object-oriented Simulator of Marine ecOSystem Exploitation) is a spatial multispecies individual-based model that uses simple individual predation rules to model trophic interactions of piscivorous fish (Shin and Cury, 2001; Shin *et al.*, 2004). It can be used to explore ecosystem effects of fisheries and to test the hypothesis that predation is opportunistic, depending only on size suitability and spatial co-occurrence between predators and their prey, as well as to help understand the (Shin *et al.*, 2004).

OSMOSE is initialized using Ecopath-based estimates of biomass, annual natural mortality and fishing mortality values (Shin *et al.*, 2004). As with the other multispecies models, OSMOSE requires a large number of input parameters, related to growth, reproduction and survival parameters. Top predators such as marine mammals and seabirds can be represented using an additional natural mortality term.

Shin *et al.* (2004) developed an 'OSMOSE' model for the southern Benguela ecosystem and a version for the North Sea, in which marine mammals will be included as predators, is currently being developed by Ifremer and Cefas.

7.4.2 Other IBM

INVITRO (Gray *et al.*, 2006) is a hybrid model currently used in management strategy evaluation in the coastal waters of Australia (e.g. Little *et al.*, 2006). In INVITRO, classical meta-population models are used for habitats while IBMs are used for higher trophic levels or species of conservation concern, such as whales.

Biophysical coupled models have been developed to study the interactions between cod populations and zooplankton and the influences of climate variability and change, in Georges Bank and the Norwegian shelf/Barents Sea (the northern and southern extremes of the distribution of Atlantic cod). The trophodynamic component was based on a standard bioenergetic supply–demand function, in which growth is represented as the difference between the amount of food absorbed by a larva and the metabolic costs of its daily activities. The formulation included: (i) variable composition of prey fields; (ii) effect of turbulence, swimming behaviour and satiation on encounters and ingestion of larval fish and their prey; (iii) light limitation on ingestion rates at low and at high light intensities and (iv) effects of temperature on metabolic costs, ingestion rates and growth (see ICES, 2007).

7.5 Bioenergetic models

A distinct suite of ecosystem models, based on bioenergetic and allometric reasoning, which involves parameterising a model using power functions of individual body mass, was developed by Yodzis and Innes (1992). Yodzis (1998) created a 29-species foodweb model incorporating allometric reasoning to investigate the effects of a re-

duction of fur seals on fisheries in the Benguela ecosystem. An improved bioenergetic model was constructed to describe interactions between squid, anchovy, hake and sea lions in the Patagonian shelf (Koen- Alonso and Yodzis, 2005). The Yodzis and Innes (1992) framework has also been expanded by Vasseur and McCann (2005) to incorporate temperature dependence.

Welch *et al.* (2010) used this approach to model trophic dynamics and the energy flow in the Lancaster Sound region of the Canadian Arctic. The model included primary production, zooplankton, bivalves, arctic cod, marine mammals and seabirds. Buren *et al.* (2014) implemented a multispecies bioenergetic-allometric model for the eastern Scotian Shelf, Labrador and Newfoundland shelf. This examined the roles of fisheries, environment and harp seal (*Pagophilus groenlandicus*) predation on the recovery of the Atlantic cod (*Gadus morhua*).

7.6 Conclusions

All the models reviewed have advantages and disadvantages, as well as proponents and detractors. The main tension is between “complete” ecosystem models that are difficult to populate and run, and minimum realistic models that sacrifice detail for ease of use. In the present context, the latter have the advantage that several such models are already in use for fish stock assessment.

Models based on energy flows are generally more relevant than those focused on element flows. Other considerations, e.g. regarding use of individual-based models vs. aggregate state models, or whether models should account for allometric growth, are perhaps less central.

Where marine mammals have been included in models, it is usually to account for their predation on fished species, in models which can (or could) in principle be used to identify and quantify resource competition and fisheries. However unless the models include realistic assumptions about prey selection, their conclusions about the effect of such competition may be erroneous.

It is clearly fundamental that we have good dietary data on marine mammals. There are both scientific and ethical-legal-political arguments as to whether it is more appropriate to use data from stranded and bycaught animals or to undertake a programme of lethal sampling. Within the EU, the latter is not an option whereas, at least in the case of abundant marine mammal species, the NAMMCO countries are likely to favour lethal sampling. However, as suggested above, a good quantitative but static picture of the diet is insufficient. Statistical modelling based on some combination of stomach contents data, tagging data on animal movements, and information on fish distribution and abundance, can potentially yield evidence about univariate and multivariate functional responses.

Biochemical indicators of diet (e.g. stable isotope ratios, fatty acid compositions) are potentially available from many more animals (e.g. stranded animals with empty stomachs and biopsy samples from living animals). However, the value of such data is still limited by the difficulties in interpreting the biochemical signals in terms of detailed and quantitative diet composition. This problem is exacerbated due to (i) different prey species, especially closely related species, having similar isotopic or fatty acid profiles, (ii) changes in profiles over an individual’s life, (iii) the need to filter out signals related to habitat or latitude rather than to diet, and (iv) differences between the biochemical signals of predator and prey tissues, and between different tissues of the same predator (i.e. “fractionation” in relation to isotope ratios and synthesis, transformation and differential absorption of different fatty acids).

It should be mentioned that there are other types of biological interaction between marine mammals and commercially exploited finfish and shellfish, for example in relation to the transmission of pathogens, parasites (e.g. anisakids) and contaminants (e.g. persistent organic pollutants). Indeed, these interactions also potentially impact on the human consumers of the fished species.

Finally, there is also a need to include operational interactions between marine mammals and fisheries in ecosystem models. Fortunately this is relatively straightforward in that bycatch rate is likely to be a simple function of gear type and fishing effort, although there may also be spatial and temporal variation related to the marine mammals' habitat requirements and life cycle. Lassalle *et al.* (2012) included both prey competition and fishery bycatch in her EwE model of the Bay of Biscay.

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Annex 2: Agenda

The Council Room, Zoological Society of London main building, Regent's Park, London, UK

Monday, 9th February 2015

09:00 Welcome and start of meeting

Paul Jepson, Rob Deaville, Institute of Zoology

9:15 Plenary session: setting up of Internet connections, adoption of agenda; discussion of ToRs, setting up of work plan

10:30 Coffee break

11:00 Continue discussion of ToRs

13.00 Lunch break

14:00 Paul Jepson talk on contaminants

15.30 Coffee break

15.45 Work in subgroups

16:45 Plenary session: discussion of work plan for next day

17:30 Close

Tuesday, 10th February 2015

09:00 Plenary session: review progress and work plan for the day

09.30 Work in subgroups

10:30 Coffee break

11:00 Continue discussion of ToRs

13.00 Lunch break

14:00 Work in subgroups

15.30 Coffee break

15.45 Work in subgroups

16:45 Plenary session: review progress and discussion of work plan for next day

17:30 Close

Wednesday, 11th February 2015

09:00 Plenary session: review progress and work plan for the day
09:30 Work in subgroups
10:30 Coffee break
11:00 Continue discussion of ToRs
13:00 Lunch break / Necropsy at IOZ
15:00 Discuss OSPAR request
17:30 Close
19:30 Dinner (see e-mail)

Thursday, 12th February 2015

09:00 Plenary: Discussion of recommendations and future ToRs
10:30 Coffee break
11:00 Continue Plenary session
13:00 Lunch break
14:00 Completion of work on ToRs
15:30 Coffee break
15:45 Completion of work on ToRs
17:30 Close

Annex 3: WGMME terms of reference for the next meeting

2015/X/ACOMXX The **Working Group on Marine Mammal Ecology** (WGMME), chaired by Begoña Santos* (Spain) and Graham Pierce* (UK), will meet in Madrid, 8–11 February 2016:

- a) Review and report on any new information on population abundance, population/stock structure and management frameworks for marine mammals;
- b) Update North Atlantic information on negative and positive ecological interactions between grey seal (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*)/other marine mammals;
- c) Update the database for seals;
- d) (OSPAR request; details to be confirmed)

WGMME will report by 4 April 2016 for to the attention of the Advisory Committee.

Supporting Information

PRIORITY	
Scientific justification	<p>ToR a) This may contribute to the work required for the MoU between the European Commission and ICES to “provide new information regarding the impact of fisheries on other components of the ecosystem including small cetaceans and other marine mammals...” and to aid Member Countries in “scientific and technical developments in the support of the Marine Strategy Framework Directive (MSFD), such as by designing marine monitoring and assessment programmes, identifying research needs and methodologies advice”.</p> <p>ToR b) There is increasing evidence of possible fatal interactions between grey seals and other (smaller) marine mammal species, particularly as the population of grey seals in the ICES area increases (while in some areas harbour seal populations are decreasing).</p> <p>ToR c) A standing requirement to keep the new seal database up to date and to ensure its usefulness.</p> <p>ToR d) tbc</p>
Resource requirements	None
Participants	The Group is normally attended by 10–20 members.
Secretariat facilities	None apart from the usual support in completing and publishing the report
Financial	No financial implications.
Linkages to advisory committees	The work of WGMME may support development of the ecosystem overviews and the new format of fisheries advice
Linkages to other committees or groups	
Linkages to other organizations	OSPAR

Annex 4: Recommendations

WGMME notes that EGs are asked not to address recommendations to “ICES” or “ICES Member Countries”. However, the majority of recommendations below concern recommendations in support of Member State conservation obligations that are currently not being met, thereby also facilitating delivery of advice by ICES. Therefore, although we address these recommendations to ACOM in the first instance, the intent is for them to be communicated to the competent authorities in the ICES Member Countries.

RECOMMENDATION	ADDRESSED TO
1. WGMME (2014) strongly supported the proposal for a cetacean absolute abundance survey in all European Atlantic waters in 2016 and recommended that it is supported by all range states and by ICES, ASCOBANS and the European Commission. Continuation of these surveys is essential to the accurate estimation of absolute abundance for several species that are required for reporting under the Habitats Directive and the Marine Strategy Framework Directive. WGMME notes the urgent need for a new large-scale absolute abundance survey for cetaceans in European Atlantic waters, following those carried out in 1994 and 2005–2007, to fulfil EU MS obligations under the MSFD and Habitats Directive. WGMME therefore recommends that all competent authorities support the implementation of the SCANS III initiative.	ACOM, for dissemination, e.g. to relevant ICES clients and other competent authorities
2. WGMME further notes the need for equivalent surveys in other European Seas covered by the MSFD where no large-scale cetacean abundance surveys have been undertaken. There is an ongoing initiative under the auspices of ACCOBAMS to carry out a cetacean survey in the Mediterranean and Black Sea. The results from such a survey would be complementary to SCANS III and would help MS meet MSFD and Habitats Directive requirements in that region. WGMME therefore recommends that all competent authorities support the implementation of such a survey.	ACOM, for dissemination, e.g. to relevant ICES clients and other competent authorities
3. In the case that no compelling reason arises for an immediate update, WGMME recommends that the “threat matrix” is updated every three years.	ACOM
4. WGMME notes that POPs such as PCBs and BFRs pose a significant threat to the status of marine mammal populations. POPs are capable of causing immune and reproductive failure and currently occur at concentrations which would be expected to cause such effects in a number of marine mammal species in European seas. WGMME recommends support for ongoing monitoring of POP concentrations in marine mammals and the inclusion of such information within (a) common indicator(s) under MSFD descriptor 8. Species to be monitored should include, but not be restricted to, harbour porpoise, seals, bottlenose dolphins and killer whales.	ACOM, for dissemination, e.g. to relevant ICES clients and other competent authorities
5. Given the emerging strong evidence of increasing grey seal predation on other marine mammals, WGMME recommends organisation of a dedicated workshop with the aim of synthesising available data and developing a coordinated research plan.	ACOM, to seek suitable organisers for such a workshop

RECOMMENDATION	ADDRESSED TO
6. OSPAR will undertake an Intermediate Assessment of common indicators for marine mammal distribution and abundance under MSFD Descriptor 1 in 2017. Related to this, it has requested assistance from ICES WGMME. Assessments foreseen will require information on historical and recent status and trends for minke whales, coastal bottlenose dolphins and other cetacean species, where relevant, and abundance data, by management unit, for grey and harbour seal. WGMME therefore recommends that the competent authorities in the OSPAR Maritime Area should provide relevant distribution and abundance data, upon specific request by OSPAR and well before the ICES WGMME meeting in February 2016.	ACOM, for dissemination, e.g. to relevant ICES clients and other competent authorities