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

6–9 February 2017

St Andrews, Scotland, UK

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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Contents

Executive Summary

The **Working Group on Marine Mammal Ecology** (WGMME), chaired by Begoña Santos (Spain) and Graham Pierce (Spain), met in St Andrews, UK 6–9 February 2017. It reported on recent information on status of, and threats to, marine mammal populations and briefly reviewed current knowledge of effects of plastics and underwater noise. Direct interactions between seals and fisheries were reviewed and the group also reported on the current status of the ICES / OSPAR seal database(s). The group provided text for five ecosystem overviews. Criteria for assessment of abundance trends in offshore cetaceans in the context of the MSFD were reviewed, modifying the proposed indicator (previously based solely on the rate of decline) to make specific reference to baseline values. Linked to this, the group reported on the outcomes of the 2016 SCANS III survey. Given that the three main large-scale surveys of cetaceans in European Atlantic waters have all arisen from individual projects and were separated by intervals of eleven years, there is concern as to the future and utility of these surveys. WGMME recommends that the surveys be co-organised and coordinated by Member States as part of their routine monitoring and that the frequency is increased to once every six years to match the MSFD reporting cycle.

1 Introduction

The Working Group on Marine Mammal Ecology (WGMME) met at the University of St Andrews (St Andrews, UK, during 6–9 February 2017. 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 University for hosting the meeting.

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, and over the last three years.

The WGMME proposes new chairs and updated ToRs for 2018 (see Appendix X) and discussed meeting venues. The University of La Rochelle (France) offered to host the 2018 meeting while Abbo van Neer agreed to explore the possibility of holding the 2019 meeting on Helgoland (Germany).

2 ToR A. Review and report on any new information on population abundance, population/stock structure, management frameworks (including indicators and targets for MSFD assessments), and anthropogenic threats to individual health and population status (e.g. plastics)

This term of reference is framed slightly more broadly than in previous years and we therefore include some historical background material on some threats not previously covered as well as providing an update for the last year. Compilation of material for the ecosystem overviews has also given rise to new information not previously reported to WGMME and which is therefore included here.

2.1 New abundance information

2.1.1 Seals

The most recent survey data on abundance of grey seals and harbour seals known to WGMME members are summarised in Tables 2.1 and 2.2 below.

Table 2.1. Recent grey seal survey data.

Country		Recent Survey Year(s)	Pups	Adults (moult)	References
Norway					
	Tomso & Finmark	2015–2016	271		Nilssen and Bjørge (2016); Oigard *et al*. (2012)
	Norway north of 62N	2014–2015	318		See [1] below
	Norway south of 62N	2008	43		
Iceland		2012		4200	Hauksson *et al*. (2012)
Wadden Sea		2016	1113	4936	Brasseur *et al*. (2016)
France		2016	43		See [2] below; Vincent *et al*. (in revision)
United Kingdom					
	Inner Hebrides	2012	4088		SCOS, 2015
	Outer Hebrides	2012	14 136		
	North Sea	2012	28 136		
	Scotland total	2012	50 025		
	England & Wales	2012	6863		
Republic of Ireland		2012	2100		Ó Cadhla *et al*. (2013)
Canada					
	Sable Island	2010	62 000		Bowen *et al*. (2010)
	Gulf of St Lawrence + eastern shore Canada	2010	14 200		Thomas *et al*. (2011)
USA	*USA east coast*	2008	2600		NOAA (2009)
Sweden					
	Skagerrak				
Baltic	Baltic	2014	33 000		Härkönen *et al*. (2013)

Table 2.2. Recent harbour seal survey data.

Country		Survey Year(s)	Adults (moult)	Pups	References
Norway					Nilssen and Bjørge (2016)
	North of 62N	2012–2015	5267		
	South of 62N	2011–2016	1128		
	Finmark	2012–2013	981		
	Skagerrak	2015–2016	638		
Iceland		2016	7652		report reference to follow
Wadden Sea		2016	24 339	7566	Galatius *et al*. (2016)
France		2016	865	130	See [2] below; Vincent *et al*. (in revision)
United Kingdom					
	Scotland	2011–2015	25 355		SCOS (2015)
	England and Wales	2007–2014	4806		
	Northern Ireland	2007–2011	948		
Ireland		2011–2012	3489		SCOS (2015)
USA		2012	75 834		SCOS (2015)
Canada					NAMMCO
	south of Labrador	1970s	12 700		
	Estuary and Gulf of St Lawrence	1994–2000	4000–5000		
Sweden and Denmark					NAMMCO
	Skagerrak	2015	6000		
	Kattegat/ Danish Straits	2015	10 000		
	southern Baltic	2015	1000		
	Limfjord	2015	1000		
	Kalmarsund	2015	1000		

FRANCE: Grey and harbour seals in France are at the southern limit of the two species' ranges. In 2016, 43 grey seal pups and 130 harbour seal pups were counted. The harbour seal counts during the moulting season (August) gave a maximum total number of 865 seals. Since the 1990s, the maximum harbour seal number has increased by 10% per year on average in the central English Channel, and by 31% per year in the eastern English Channel. Grey seal counts in summer (August) and the moulting season (March) gave maximum numbers of 810 and 720 grey seals respectively, along the French coasts. The average annual rate of increase of grey seals varies significantly between regions, being 6–8%/year in colonies in the western English Channel as compared to 49%/year in the northeastern Channel. This rapid rate of increase in grey seal numbers in northern France is probably due to immigration of animals from the North Sea (Vincent *et al*., in revision).

ICELAND: Icelandic harbour seal and grey seal populations are currently in decline. The harbour seal population decreased from 33 000 animals in the first census in 1980 to 7700 animals in 2016. The largest observed decline, however, occurred between 1980 and 1989 when a bounty system was in effect, but the declining trend continues and the current estimated population size is the smallest that has ever been observed. The Icelandic grey seal population has been surveyed at irregular intervals since 1982 when the population size was estimated at 9000 animals. The latest estimate from 2012 indicated a population size of 4200 animals. A new grey seal census is planned in 2017.

NORWAY: A 2012 survey of harp and hooded seals in the Greenland Sea showed stable levels for these two species, with harp seals at a high level and hooded seals at a continued historical low level. Surveys of grey seals along the Norwegian coast indicate a reduction in pup production by between 50–60% between 2007–2008 and 2014–2015 in mid-Norway. The abundance of harbour seals in central Norway has also declined since the late 1990s, mainly because of hunting, but the population is at present recovering. The decline in the grey seal population is probably mainly due to increased bycatches in gillnet fisheries for monkfish and cod.

2.1.2 Cetaceans

BELGIUM: The harbour porpoise (*Phocoena phocoena*) is the most abundant cetacean in the Belgian part of the North Sea (BPNS). Aerial surveys revealed that average densities in these waters range from 0.2 to 4 animals/km² (Haelters *et al*., 2013a; 2015). In 2016, two aerial surveys were conducted covering the Belgian part of the North Sea. In mid-April and mid-June average density estimates were, respectively, 1.2 (0.9–1.7) and 0.9 (0.5–1.5) animals/km². The apparent decrease in density between April and June may in part be due to disturbance from hydraulic piling. A deterrent effect across distances of ~20km following pile-driving was previously observed in Belgian waters by Haelters *et al*. (2013b). Offshore construction for the Nobelwind windfarm started in 2016 with hydraulic piling taking place from May to September 2016.

Haelters *et al*. (2016) analysed data from static passive acoustic monitoring (PAM), collected between 2010 and 2015 at two locations using c-PoDs. They found a significant seasonal trend in detections, assessed by month, with peaks in late winter–early spring and late summer, consistent with the results of aerial surveys and with strandings data (Haelters *et al*., 2016a).

In 2016, strandings of harbour porpoises in Belgium increased again after a steep decline in 2015 (Figure 2.1; Haelters and Geelhoed, 2015). The major causes of death were bycatch, and direct or indirect mortality (infected wounds) as a result of predation by grey seals (*Halichoerus grypus*). These each account for more than 30% of deaths in cases where cause of death could be determined (Haelters *et al*., 2017). In recent years, increased incidences of scavenging on stranded porpoises by red foxes (*Vulpes vulpes*) have made it more difficult to identify the cause of death (Haelters *et al*., 2016).

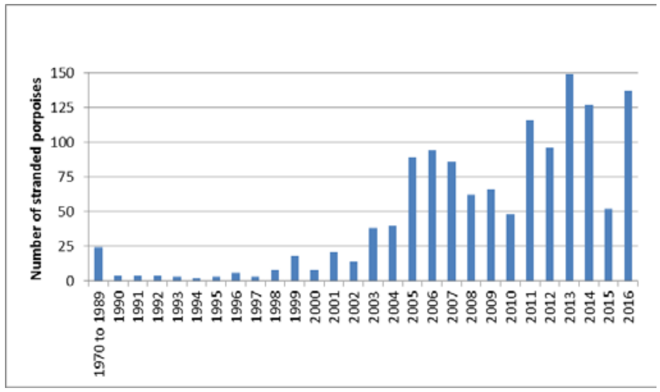


Figure 2.1. Strandings of harbour porpoises in Belgium recorded annually from 1990 to 2016 (plus total for 1970–1989). Data from Haelters *et al*. (2017).

FRANCE: The recurrent cetacean and seabird sighting programmes conducted on board RV Thalassa during the fish stock assessment surveys PELGAS, IBTS, CGFS and EVHOE have continued during 2016 and will do so in 2017. No specific survey dedicated to estimating cetacean abundance and distribution was conducted in 2016. Three papers dealing with cetaceans in French waters of the Northeastern Atlantic are in press in a special issue of Deep-Sea Research-Part II. These studies are based on data collected during two large dedicated aerial surveys named SAMM (*Suivi Aérien de la Mégafaune Marine*, Aerial Census of Marine Megafauna). These surveys were conducted over the Bay of Biscay and English Channel during winter 2011–2012 (late November to mid-February; 32 433 km of sampled transects) and summer 2012 (mid-May to early August; 33 864 km of sampled transects) and allowed a seasonal comparison of the abundance and distribution of cetaceans in the Bay of Biscay and the English Channel (Laran *et al*., 2017). The most abundant species encountered in the Channel, harbour porpoise, displayed strong seasonal variations in its distribution but a stable abundance (18 000 individuals, CV=30%). In the Bay of Biscay, both abundance and distribution of common/striped dolphins varied seasonally, with 285 000 individuals (95% CI: 174 000–481 000) in winter, preferentially distributed close to the shelf break, and 494 000 individuals (95% CI: 342 000–719 000) distributed beyond the shelf break in summer. Seasonal abundances of bottlenose dolphins were quite stable, with a large number of ‘pelagic’ encounters offshore in winter. Finally no significant seasonal difference was estimated for pilot whales and sperm whales (Laran *et al*., 2017).

Lambert *et al*. (2016a) explored how ocean seasonality drives habitat preferences, using Generalised Additive Models to determine relationships with physiographic variables (depth, slope), and weekly and monthly averaged oceanographic predictors (mean, variance and mean gradient of sea surface temperature; mean and standard deviation of sea surface height; maximum tidal velocity), for both seasons. Long-finned pilot whales and Risso’s dolphins exhibited no habitat variations between seasons, targeting the shelf break throughout the year whereas harbour porpoises, common and striped dolphins, and bottlenose dolphins all modulated their habitat preferences between seasons (Lambert *et al*., 2016a). These models were also used to assess the networks of existing Natura 2000 sites, for winter and summer independently, and proposed offshore areas of biological interest (Lambert *et al*., 2016b). The results showed a clear shortfall for cetaceans in the Atlantic region but indicated that the proposed large offshore areas of interest would constitute a highly relevant network for all offshore species, with *e.g.* up to 61% of the Globicephalinae population in the Atlantic French waters being present within these areas.

ICELAND: Cetacean surveys conducted at regular intervals between 1987 and 2016 have revealed varying trends in abundance. Humpback whales have shown high rates of increase and fin whale abundance also increased significantly during 1987–2001. Abundance of common minke whales has decreased substantially in Icelandic coastal waters since 2001, most likely due to decreased availability of important prey species such as sandeel (Ammodytidae) and capelin (*Mallotus villosus*).

NORWAY: The most recent counting cycle of NEA minke whales for 2007–2013 shows a stable overall population level but a general displacement of minke whales and other baleen whales towards the Northeast, implying a shift from the Norwegian Sea to the Barents Sea.

SPAIN: Trends in abundance of the common dolphin (*Delphinus delphis*) in shelf northwestern Spanish waters: Along the north and northwest coasts of the Iberian Peninsula (ICES subareas 9.aN and 8.c), the Spanish Institute of Oceanography (IEO) has carried out annual acoustic surveys (PELACUS) to estimate pelagic fish biomass for the last two decades. Since 2007, an observer programme of top predators has been integrated into the surveys, collecting sightings on cetaceans, seabirds and other species using line-transect methodology in passing mode with a single platform configuration. The absence of a double platform precludes correction for animals missed on the track-line. Thus, if cetaceans are not sighted before they respond to the ship, when animals are attracted to the ship, abundance will be overestimated (Mullin and Fulling, 2003). While absolute abundance cannot be estimated, we assume that the attraction or evasion movements exhibited by certain species of cetaceans are maintained over time if the same platform and method of sampling are used, and therefore the positive or negative bias on abundance estimates would also not change. The proportion of individuals of a given species missed on the track-line would also remain constant if the conditions and experience of observers also do so. Therefore, although the absolute estimates may be biased, the trends we detect will still be informative.

Common dolphin sightings from 2007 to 2016 were analysed using a conventional design-based line transect methodology (Buckland *et al*., 2001) with multiple covariates (Marques and Buckland, 2003) using the standard software Distance version 6.2 release 1 (Thomas *et al*., 2010).

The study area is the continental shelf waters belonging to the Northwest Spanish subregion, as defined under the Marine Strategy Framework Directive (MSFD; Directive 2008/56/EC 2008), with a surface of ≈37 000 km^2. The survey design consists of a systematic grid within this area with linear transects perpendicular to the coastline equally spaced (8 nautical mile apart) covered at 10 knots/hour. Transects performed inside the rías (i.e. coastal inlets) were excluded from the analyses because of their very different environmental and habitat characteristics. In some years, the sampled area was slightly extended into adjacent Portuguese and French waters, and/or additional transects within the study area were surveyed, using the same methodology.

All sightings under Beaufort conditions ≤ 5 and within a truncation distance of 650 m (N=150; **Table 2.2.1**) were used to fit the Detection Function (DF) but only those sightings belonging to the predefined transects inside the study area were used to calculate encounter rates and to estimate abundances and densities. The best model for the DF (based on AIC values) with a truncation distance of 650 m had a half-normal key function, with Beaufort and log of the cluster size as covariates according to the lower AIC value among the total combinations of covariates tested, with an effective strip half width of 230.35 m (CV 0.07).

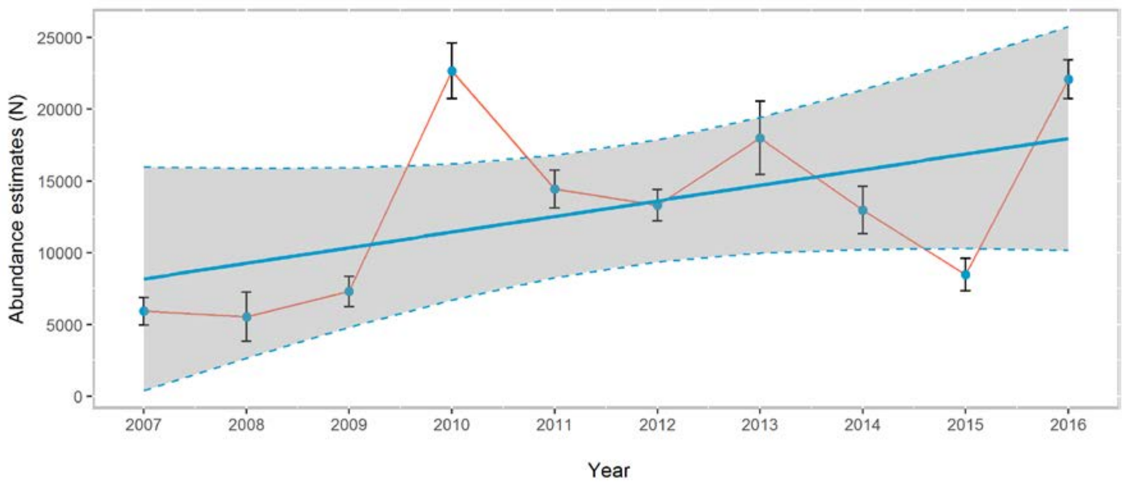
Table 2.1.2.1. Number of sightings (N) with Beaufort ≤5 and truncation distance ≤650 m (B≤5 T650 m) used for Detection Function selection; only within the study area and predesigned transects (B≤5 SA T650 m); and Kilometres (Km) with Beaufort ≤5 within the study area and predesigned transects (B≤5 SA).

	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	Total
N (B≤5 T650 m)	11	12	11	20	15	17	15	16	9	24	150
N (B≤5 SA T650 m)	7	4	8	18	13	17	14	13	9	24	127
Km (B≤5 SA)	1555.1	952.0	1442.1	1046.0	1185.8	1680.5	1024.6	1318.8	1398.5	1431.1	13 034.5

The DF estimated with the overall sightings performed with Beaufort ≤ 5 was applied to the predefined transects inside the Spanish study area (total length 13034.5 km) and to the 127 sightings of common dolphins within the truncation distance (**Table 1**). The estimated mean group size was 16.6 (CV 0.12), mean abundance over the whole period was 12831 dolphins (CV 0.18) and mean density was 0.35 animals/km^2 (CV 0.18). Annual abundance estimates ranged from 5533 (density 0.16; CV 0.62) in the second year (2008) to 22662 (density 0.61; CV 0.36) in 2010, with an overall slight upward trend (Figure 1).

A linear regression fitted to estimated annual abundance (N) vs. year (Yr, the position of the year in the time-series, from 1 to 10) had a positive slope but was not statistically significant (P=0.124). However, the uncertainty around each estimate is not taken into account when fitting a regression in this way. Therefore, using the mean and CV of abundance values for each year and assuming a lognormal distribution of errors, 1000 simulated datasets were generated from these data and each was tested for the existence of a trend. This process yielded negative trends in 1.7% of cases (none of which were individually significant), while 98.3% yielded a positive trend of which 10.4% were individually significant. The fact that over 95% of simulations showed a positive trend could be viewed as indicating a statistically significant upward trend in abundance (Saavedra *et al.*, Accepted).

(a)



(b)

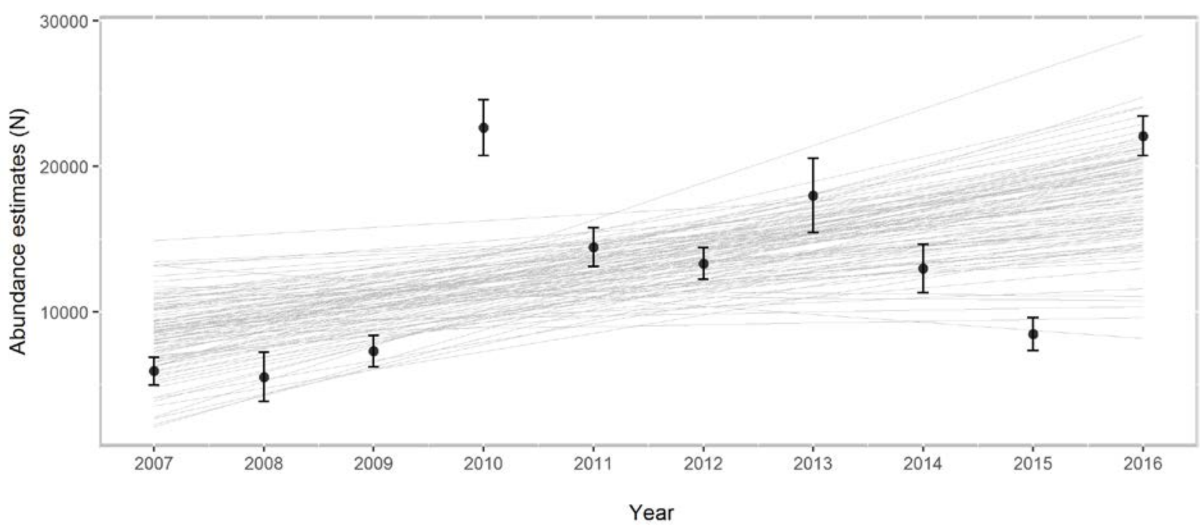


Figure 1. (a) Time-series of common dolphin estimated abundance (blue dots) in the Northwest Spanish subregion shelf waters, with standard errors (SE bars). Linear regression (blue line) fitted to annual estimates, with 95% CI (dashed lines and grey area). (b) The annual estimates plus simulated trends accounting for error in the annual abundance estimates.

UK: Line-transect surveys of Cardigan Bay in summer 2016 estimated harbour porpoise abundance at 828 (CV=0.19) and bottlenose dolphin abundance at 289 (CV=0.23) individuals (Lohrengel and Evans, 2017). Closed population estimates in summer 2016 from Capture–Mark–Recapture Photo ID analysis were 147 (95%CI=127-194) in Cardigan Bay SAC, and 174 (95%CI=150-246) in the wider Cardigan Bay for bottlenose dolphins (Lohrengel and Evans, 2017). Bottlenose dolphin birth rates in 2016 were 2.87% (based on closed population estimates), the lowest since the study started in 2001.

2.2 New information of stock structure

UK: Resident communities of bottlenose dolphins (*Tursiops truncatus*) occur in coastal waters of western Scotland. In-depth research by Van Geel (2016) has confirmed results from previous preliminary photo-ID studies (2006–2007 data; Thompson *et al*., 2011; Cheney *et al*., 2013) and genetic data (Islas-Villanueva, 2009), and provided evidence of the continued (at least 2006–2013) presence of two small, geographically and socially isolated communities on the west coast of Scotland. The Inner Hebrides (IH) community contains approximately 20 resident individuals (identified in ≥8 years since 2001), ranging across the entire western coastline of mainland Scotland out to the northern Outer Hebrides. The Sound of Barra (SoB) community contains approximately 15 individuals and resides almost exclusively in the Sound of Barra, in the southern Outer Hebrides. These communities show prolonged (at least since 2006) social and geographic segregation and contrasting ranging behaviour. Studies of both communities revealed long-term site-fidelity and year-round residency. The SoB group appears dominated by females, which cannot be attributed to dispersal by males or higher male mortality. Despite a lack of full genetic isolation, strong genetic population differentiation has been found between samples from the Scottish west coast and those from the better-studied bottlenose dolphin population in eastern Scotland (Parsons *et al*., 2002; Islas-Villanueva, 2009).

Considering the prolonged social and geographic isolation between both Scottish west coast communities revealed by Van Geel (2016), these communities appear to represent separate biological units and could therefore be considered independent management units. Whereas the SoB community and other dolphins inhabiting coastal waters off western Scotland were initially proposed as two independent management units (Evans *et al*., 2009; ICES, WGMME 2012; Cheney *et al*., 2013), recent assessments have combined these into one management unit for species management in UK waters, the coastal West Scotland and Hebrides (CWSH) unit (IAMMWG, 2015). Recent evidence of spatial and social isolation supports the earlier recommendation of the SoB community being treated as a separate assessment unit for management of bottlenose dolphins in Scottish waters. Genetic studies would be beneficial to investigate whether these two communities also represent discrete demographic entities.

2.3 New information on anthropogenic threats

2.3.1 Fishery bycatch

FRANCE: The French National Stranding Network (*Réseau National d'Echouage, RNE*) is the main tool for monitoring marine mammal stranding. External examination of the stranded individuals allows identification of bycatch marks that indicate incidental capture in fishing gear. Table 2.3.1 shows the number of individuals of each species examined during the last three years that exhibited capture marks (Van Canneyt *et al*., 2014; 2015; 2016).

Considering the Phocidae, 36 individuals of grey seals exhibited capture marks on 220 examined during the last five years, *i.e.* 16%, and 24 individuals of harbour seals on 156 examined, *i.e.* 15%. Among the best represented cetacean species in the strandings (common dolphins, harbour porpoises and bottlenose dolphins), the percentage of individuals exhibiting incidental capture marks ranged from 20% for bottlenose dolphin in 2013 up to

52% for common dolphins in the same year. In the framework of the MFSD, an alternative bycatch estimate from stranding records has been developed in France (Peltier *et al*., 2016).

Table 2.3.1. Number of individuals and number of incidental catch detected after external examinations during years 2013, 2014 and 2015. (* indicates species for which Mediterranean individuals are included).

	2013			2014			2015		
	Dead stranding	**External examination**		**Dead stranding**	**External examination**		**Dead stranding**	**External examination**	
Species	Number of individuals	Number of external examinations	By-catch marks	Number of individuals	Number of external examinations	By-catch marks	Number of individuals	Number of external examinations	By-catch marks
Cetacea	8	6		3			9	6	1
Balaenopteridae	1	1					2	2	-
Balaenoptera acutorostrata	2	2					5	5	-
Balaenoptera physalus	5	2		4	6		4	4	-
Megaptera novaeangliae							1	1	-
Odontoceti	30	16	1	28	13	1	11	7	1
Delphinidae	91	42	6	95	27	10	46	19	1
Delphinus delphis	476	470	248 (52%)	409	368	172 (47%)	221	201	90 (45%)
*Stenella coeruleoalba**	79	71	17	70	68	21	68	65	15
*Tursiops truncatus**	39	39	8 (20%)	49	47	17 (36%)	41	36	10 (28%)
Lagenorhynchus albirostris							1	2	-
*Globicephala melas**	17	16	4	17	17	5	6	16	-
*Grampus griseus**	3	2		4	5	1	6	**7**	-
Orcinus orca							1	1	-
Phocoena phocoena	487	455	170 (37%)	301	283	126 (44%)	186	163	61 (37%)
Kogia breviceps		1		1	2	1	1	3	-
Kogia sima							-	1	-
Physeter macrocephalus				4	3		3	-	-
Ziphius cavirostris	1	1		4	4		2	3	1
Mesoplodon bidens	1	1							
Phocidae	14	12	2	20	8		26	18	2
Halichoerus grypus	34	61	3	85	155	12	40	61	6
Phoca vitulina	37	74	6	38	77	4	48	78	9
Total	1325	1272	465 (36%)	1 132	1083	370 (34%)	728	699	197 (28%)

GERMANY: Culik *et al*. (2016) describe tests of the Porpoise Alerting Device (PAL), an alternative to the regular pinger, which produce aggressive click train types, in the Baltic Sea. It is unclear if this is approach will be transferable to other regions. The PAL emits signal at 133 kHz and SPLrms of approx. 151 dB re 1μPa, with a repetition rate of 20s (1.2s signal length). Sound propagation is less than for other pingers due to high transmission loss for high frequencies.

Further investigations are needed to test if this device really leads to a better detectability of nets and an associated increase in attention or is only a deterrent. Investigations with net-rows, PAL signals and simultaneous recordings of echolocation behaviour are needed. The effects on harbour porpoise communication signals also have to be further investigated. Despite these uncertainties, in May 2017, 1500 new PAL devices were provided by the government for German set-net fishermen as a tool to prevent incidental bycatches of harbour porpoises.

UK: Entanglement in static fishing gear, especially shellfish creels (pots), is a known source of mortality and injury for humpback whales (*Megaptera novaeangliae*) in the UK, the frequency of which seems to have increased recently in Scottish coastal waters (Ryan *et al*., 2016). Of the 213 incidental sighting records from 1992 to 2016, 5.6% (n = 12) comprised known entanglements. For the five most recent years (2012 to 2016), this proportion was slightly higher, at 7.5% (n = 10). Over half of the known entanglements (n = 7) involved creels, three others involved ropes which could have come from creels, and one involved an aquaculture (salmon) pen. Rescue responses to six of the 12 entangled whales resulted in successful disentanglements, although long-term survival of the animals remains unknown. Three of the entanglements (i.e. 25%) were fatal. Given the relatively small population size of humpbacks in UK waters, despite increasing numbers of sightings, there is concern that mortality from bycatch may be unsustainable for this population (Ryan *et al*., 2016).

2.3.2 Pollution: persistent organic pollutants and toxic elements

GENERAL: New pan-European collaborative research on persistent organic pollutants (POPs) shows that Europe still has a major problem with persistent polychlorinated biphenyls (PCBs) (Jepson *et al*., 2016). In this study several European cetacean species had very high mean blubber PCB concentrations, at levels likely to cause population declines and suppress population recovery. In a large pan-European meta-analysis of stranded (n=929) and biopsied (n=152) cetaceans, three out of four species considered (bottlenose dolphins, striped dolphins and killer whales) had mean PCB levels that markedly exceeded all known marine mammal PCB toxicity thresholds. Some locations (e.g. western Mediterranean Sea, southwest Iberian Peninsula) are global PCB "hot spots" for marine mammals. Blubber PCB concentrations have now stabilised in some cetaceans in the NE Atlantic, and concentrations in harbour porpoise are below the estimated threshold for effects. However, some small or declining populations of bottlenose dolphins and killer whales in the NE Atlantic were associated with low recruitment, consistent with PCB-induced reproductive toxicity.

In a global review of marine apex predators, the killer whale remains the most PCB-contaminated mammalian species and is likely to be impacted throughout its range (Jepson and Law, 2016). Other species or populations considered under potential or serious threat from PCBs include false killer whales (*Pseudorca crassidens*), ringed seals (*Pusa hispida*) in the Baltic Sea, all marine mammal species in the Mediterranean and Black Seas, beluga (*Delphin apterusleucas*) in the Saint Lawrence River (Canada) and polar bears (*Ursus maritimus*) in the Arctic.

Despite regulations and mitigation measures to reduce PCB pollution, their legacy in marine foodwebs continues to be of concern. Two of the remaining killer whale populations in the NE Atlantic, the Strait of Gibraltar population and the west of Scotland/Ireland population, are already threatened.

Two recent papers investigated mechanisms for detoxification of heavy metals in long-finned pilot whales. Pelagic odontocetes are exposed to naturally high levels of heavy metals in the marine environment and have well-developed metabolic pathways to detoxify or sequester harmful compounds of, for example, mercury (Hg) and cadmium (Cd). Detoxification of Hg relies on formation of mercury-selenium complexes, and selenium (Se) is thus protective against the toxicity of Hg. It was found that these detoxification mechanisms are fully developed from an early age. As a consequence of Hg detoxification, the selenium pool in the system is used up, and it was shown that the bioavailable Se pool could become depleted if Hg exposure levels were high, a plausible driving mechanism of demonstrated neurotoxic effects of MeHg in organisms affected by high dietary Hg intake. Although significant gaps remain in the understanding the mechanism of Hg detoxification, it appears that, whilst cetaceans seem to tolerate environmental levels of heavy metals, this capacity can be saturated in areas of high exposure, such as areas with significant anthropogenic input into the marine environment (Gajdosechova *et al*., 2016a, b).

GERMANY and DENMARK: A recent study of trace and mineral elements in blood samples from harbour seals from Helgoland and Anholt (Kakuschke and Griesel, 2016) found mineral elements concentrations within reference ranges and trace element concentrations similar to those reported in previous studies. However, the concentrations of some elements were significantly lower in the offshore than the inshore animals, suggesting variability of exposure to sources such as industrial activities, sewage, shipping traffic and dredging operations.

GREENLAND: (Levin *et al*., 2016) measured blubber PCBs in East Greenland ringed seals and found that they were within the range reported in this population in 2004 and well below the threshold for physiological effects. Concentrations of perfluoroalkyl substances (PFAS) in the serum were also low. Concentrations of contaminants in the blubber and were not significantly correlated with immune function or lymphocyte proliferation assays. The authors conclude that these ringed seals are not currently at risk of immune effects from exposure to POPs or PFAS.

PORTUGAL: Trace elements were studied in harbour porpoises from Portugal between 2005 and 2013 in relation to sex, body length, nutritional state, presence of parasites and gross pathologies. Within European waters, porpoises stranded in Portugal had the highest mercury concentrations and the lowest cadmium concentrations, which may reflect dietary preferences and the geographic availability of these pollutants (Ferreira *et al*., 2016).

2.3.3 Plastics and other marine debris

Marine debris is defined by NOAA as any persistent solid material that is manufactured or processed and directly or indirectly, intentionally or unintentionally, disposed of or abandoned in the marine environment. Marine debris is considered to be an increasing problem for marine animals, with plastic being of particular concern. Plastic debris incapacitate, asphyxiate or starve wildlife, distribute non-native and potentially harmful organisms, absorb toxic chemicals and degrade into micro-plastics that may subsequently be ingested (Barnes *et al*., 2009) Oceanic sources of plastic include abandoned, lost or discarded fishing gear (ALDFG or ghost gear), flotsam lost overboard from shipping, effluent from industrial processes and debris washed or blown into the marine environment from land.

In 2015 an International Whaling Commission (IWC) report to ASCOBANS concluded that "*Marine debris - especially in the form of plastics - has now been widely recognised as a threat of international concern. This has created significant interest in many international fora, which have developed a range of actions in response. Liaison between the IWC and these fora is advocated and this should include the Global Partnership on Marine Litter, the Food and Agriculture Organization of the United Nations (i.e. via the Committee on Fisheries), the International Maritime Organization (i.e. via the Marine Environment Protection Committee) and the Global Ghost Gear Initiative*" (Simmonds and Toole, 2015). This work followed on from a series of IWC workshops which, among other conclusions, noted the need for a collaborative approach between relevant organisations and a coordinating body to help bring these initiatives together, in order to achieve "*consistency of approach, synergy of effort and exchange of information to develop appropriate mitigation strategies that recognise that (a) prevention is the ultimate solution but that (b) removal is important until that ideal is realised*."

Marine debris can be a problem for some cetaceans both in terms of ingestion and entanglement. Ingestion of macroplastics can interfere with digestion, occupying space in the stomach and cause physical obstruction. Many species of marine mammals have been reported to have ingested marine debris, and whilst identifiable pathology is often correlated with the volume ingested, Jacobsen *et al*. (2010) comment that even small quantities can have large effects.

Entanglements of marine mammals in marine debris can cause drowning, impaired movement, deep tissue laceration, infection and starvation. They can also represent a significant financial cost to fishermen due to loss of gear and indeed a serious safety issues for those involved in disentangling entangled animals. The IWC held workshops on large whale entanglements in 2010, 2011 and 2015 and a workshop on welfare issues in 2016 (see IWC/62/15, IWC/64/WKM&AWI REP1, IWC/66/WK-WI Rep (IWC/62/15 2010), IWC/66/WKM&WI Rep 01, 2016). The workshops covered entanglement monitoring systems and entanglement prevention, mitigation, and response programmes, as well as welfare issues. The main conclusions include:

- Given the likely substantial underreporting of entangled whales, it was recommended that coastal nations (and especially fishing nations which are members of the IWC) establish programmes for monitoring whale entanglements (and that these would then be reported to the IWC);
- Monitoring of entanglements should make use of studies of scars, interviews with fishermen and whale watching operators, as well as stranding data;
- Development of monitoring and response capabilities should include working with mammal stranding networks to expand their response capabilities (e.g. by providing training in disentanglement procedures) and make better use of existing stranding data;
- Where entangling gear could be tracked to its origin, the majority was actively fished when the whale encountered it, rather than ALDFG;
- Entanglement in fishing gear is the most significant threat to wild cetacean welfare.

Questions about the impact on plastic on marine mammals were raised over 25 years ago. Walker and Coe (1989) noted that odontocetes are discriminating feeders with well-developed echolocation skills. Therefore, ingestion might occur incidentally, or result from ill health or from presence in an atypical environment (i.e. an out of habitat individual), resulting in ingestion of abnormal items by some animals.

Baulch and Perry (2014) compiled information ingestion of debris by 462 individual cetaceans, representing nine mysticete and 39 odontocete species. Debris-induced mortality rates varied between 0% and 22% in stranded animals, suggesting that debris could be a significant conservation threat to some populations. A juvenile minke whale (*Balaenoptera acutorostrata*) found freshly dead on the beach of Nieuwpoort (Belgium) in March 2013 was the first confirmed case of fatal litter ingestion in a baleen whale. The animal was very emaciated, the size of the stomach was reduced and 400 g of compacted plastic bags was found in the gastric lumen, obstructing the junction to the third stomach (Jauniaux *et al.*, 2014).

Based on data collected by the UK Cetacean Strandings Investigation Programme, during 8200 necropsies, the overall frequency of incidental macroplastic ingestion in cetaceans and pinnipeds is low. In many of these cases the plastic was thought to have been incidentally ingested during a live stranding event. Only one case of direct mortality due to plastic ingestion is recorded in the UK CSIP strandings record, an adult male Cuvier's beaked whale. The UK dataset highlights the importance of routine collation of negative as well as positive data, to help build up a broader picture of where debris ingestion is an issue for a particular species and region, and where it is not (IWC/66/WKM&WI Rep 01, 2016).

Two species in particular have been reported to ingest large quantities of plastic: the sperm whale and Cuvier's beaked whale (*Ziphius cavirostris*). Jacobsen *et al*. (2010) found remains of up to 16 m^2 of 134 different types of net in a single sperm whale, while eight of 22 sperm whales stranded along the coasts of the North Sea in January and February 2016 had ingested marine debris comprising netting, ropes, packaging material and pieces of hard plastic, subsequently identified as fragments from a car (Unger *et al*., 2016). Poncelet *et al*. (2000) recorded 378 items with a total weight of 33 kg in a Cuvier's beaked whale. Brownlow *et al*. (2015) documented an adult male Cuvier's beaked whale stranded in Skye (UK) in December 2015, which had a severe impaction of around 4.3 kg of plastic bags and sheeting in its stomach and duodenum. There were many disease processes occurring in this animal but severe gut lesions were likely due to plastic ingestion. A Cuvier's beaked whale stranded on the island of Sotra, Bergen (Norway) in January 2017 had 30 plastic bags and several smaller plastic fragments in its stomach. The volume of plastic in the stomach was sufficient to prevent prey from being digested. Again the presence of plastic was implicated as a significant contributing factor to mortality. Small fragments of plastic have been implicated in mortality in this species, for example due to occlusion of the oesophageal sphincter in a pregnant female stranded in Brazil (Bortolotto *et al*., 2016).

Microplastics, small plastic particles (<5 mm) are attracting increasing attention as a health risk to marine organisms including marine mammal. Primary microplastics are those manufactured as small beads, for example as feedstock for the plastics industry or for use as abrasives. Secondary microplastics result from the fragmentation of larger plastic debris or from other degradation sources such as fibres in wastewater from washing clothes (Betts, 2008). In small marine organisms, microplastics can block feeding appendages and apparatus, hinder the passage of food through the intestinal tract and cause pseudo-satiation, resulting in reduced foraging and food intake (Moore, 2008). In addition, they can leach chemicals such as plasticizers (e.g. phthalates), additives (e.g. bisphenol A) and stabilisers (e.g. organotins) and adsorb water-borne contaminants (e.g. organochlorines, brominated flame retardants, chlorinated pesticides and polycyclic aromatic hydrocarbons, (Betts, 2008).

In higher marine organisms, such as seals and cetaceans, evidence of direct effects of microplastics is limited. Leached plastic additives were found in the blubber of stranded

Mediterranean fin whales (*Balaenoptera physalus*) (Fossi *et al*., 2012; 2014). The mono-(2-ethylhexyl) phthalate (MEHP) concentrations found in the blubber and in their euphausiids prey suggested that phthalates might be useful as an indicator of exposure to microplastics. However, this finding requires further research as untreated sewage could also be the source of phthalates. Experimental studies in seabirds have shown that absorbed contaminants (such as organochlorines) can be transferred and assimilated into the body from ingested microplastics (Yamashita *et al*., 2014). Nevertheless, the role of plastics in the bioaccumulation of persistent organic contaminants is probably relatively minor compared to that of naturally occurring particles and contaminated prey (Bergmann *et al*., 2015; Herzke *et al*., 2016).

BELGIUM: In April 2016 the decomposed carcass of a juvenile male narwhal (*Monodon monoceros*) was found in the river Scheldt (Belgium), 90 km upriver. The animals was very emaciated with a number of litter items (including plastic and wood, see Figure 2.3.3) in its stomach, while prey remains were completely lacking. Due to the advanced state of decomposition of the carcass, a definitively cause of death could not be established, but there was no direct evidence of fatal gastric impaction due to the plastic ingestion. The animal had been observed alive a month before the discovery of the carcass. This and other recent out of range observations of Arctic vagrants in Europe, such as bowhead whale (*Balaena mysticetus*), beluga (*Delphinapterus leucas*) and harp seal (*Pagophilus groenlandicus*) might reflect ongoing changes in the Arctic ecosystem (Haelters *et al*., in prep.).

Figure 2.3.3. The narwhal stranded in Belgium and its stomach contents.

2.3.4 Underwater noise (offshore construction, shipping, aquaculture)

2.3.4.1 Introduction and review

Growth of the offshore windfarm sector continues apace across the ICES area (EWEA, 2016). The vast majority of windfarms currently active or under construction in European waters involve the use of monopiles, which require extensive pile-driving operations along with substantial support vessel activity (Brandt *et al.*, 2016; EWEA, 2016). Noise generated during pile-driving operations can travel for long distances (e.g. Tougaard *et al.*, 2009; Brandt *et al.*, 2011) and there are long-standing concerns about the effects on marine mammals (e.g. Madsen *et al.*, 2006; Bailey *et al.*, 2010; Dolman and Simmonds, 2010).

Thompson *et al.* (2013) assessed population-level consequences of windfarm-related disturbance of harbour seals (*Phoca vitulina*) in the Moray Firth (UK), incorporating uncertainty in various crucial parameters. Although short-term responses varied, effects were estimated to be limited over longer periods (25 years; Thompson *et al.*, 2013).

Windfarm construction activities can impact on harbour porpoises over a wide area (Tougaard *et al.*, 2009; 2011; Brandt *et al.*, 2011; 2012; 2016). A deterrent effect across distances of ~20 km was observed in the Belgian EEZ by Haelters *et al.* (2013) following pile-driving efforts near the boundary with the Dutch EEZ, although it is unknown how long this effect persisted. The impact of pile-driving depends not only on the intensity of the sound but also on the duration of exposure. Kastelein *et al.* (2015; 2016) reported on experimental exposure of a captive porpoise to playback of pile-driving sounds. A Temporary Threshold Shift (TTS) in hearing sensitivity at frequencies of 4 and 8 kHz was generated after 60 minutes of exposure to a typical pile-driving sequence; hearing had recovered by 48 minutes post-exposure (on average). The risks posed by prolonged and repeated exposures (e.g. from multiple construction sites) remain poorly understood although, evidently, the possibility of permanent hearing damage is a particular concern.

Verfuss *et al.* (2016) reviewed mitigation of noise impacts associated with pile-driving operations. In UK waters, licensing conditions for site development include requirements for visual and passive acoustic monitoring of an exclusion zone (~500 m) around the pile driving site, followed by 'soft starts' and/or early shutdown of operations if marine mammals are observed within this zone. Seasonal restrictions on pile driving activities have been put in place in the Netherlands and Belgium, to avoid undue exposure of animals during the breeding season. In Belgium, emitted impulsive sounds (e.g. from pile-driving) should not exceed 185 dB re 1 μPa zero-to-peak sound pressure level (SPL) at 750 m from the source (Degraer *et al.*, 2012). In Germany, emitted impulsive sounds should not exceed 160 dB re 1 μPa 2 s (SEL) or 190 dB re 1 μPa peak to peak (SPL) at 750 m from the piling site (German Federal Maritime and Hydrographic Agency, 2010; German Federal Environment Agency, 2011; UBA, 2011).

Active mitigation methods such as bubble curtains in the water column around pile-driving operations can reduce the area over which porpoises are disturbed by noised from pile-driving by up to 90% (Lucke *et al.*, 2011; Nehls *et al.*, 2016), although this can vary according to environmental conditions (Brandt *et al.*, 2016).

There has been an increase in the use of acoustic deterrent or harassment devices (ADDs or AHDs) to exclude marine mammals from areas around pile-driving operations associated with windfarm construction (Verfuss *et al.*, 2016). These devices have also increasingly been proposed as a mitigation method for tidal turbine developments, to prevent animals colliding with the turbine blades (Coram *et al*., 2014). Wilson and Carter (2013) reviewed the efficacy of using active deterrents in tidal energy developments, identifying the key characteristics of effective acoustic deterrents and concluding that existing ADD/AHD designs were inappropriate to the specific conditions prevailing in tidal-stream sites, pointing to a need for further research. A review by Hermannsen *et al*. (2015) pointed out that the radius for absolute deterrence of harbour porpoises was only 200 m (Airmar) or 350 m (Lofitech) and that the radius for less than total deterrence was between 1.3 and 1.9 km.

AHDs and ADDs are also used in marine aquaculture, to discourage seal predation on caged fish, and in certain regions (e.g. western Scotland) may contribute significantly to overall anthropogenic noise levels (Nowacek *et al*., 2007; Shapiro *et al*., 2009). Devices used in this context have a typical peak frequency range of 5–35 kHz and source levels of >185 dB re 1 μPa @ 1 m (Taylor *et al*., 1997; Fjalling *et al*., 2006; Booth, 2010). Concerns have been raised about the impacts of this noise on both target and non-target species of marine mammals, which could include physical injury, behavioural responses leading to displacement from preferred areas, and hearing loss (Coram *et al*., 2014; Harris *et al*., 2014; The Scottish Government, 2014). As for any noise source, the severity of impacts will depend on factors such as distance from the sound source, duration of exposure, and propagation characteristics of the environment (e.g. Harris *et al*., 2014). Lepper *et al*. (2014) investigated effects of water depth, seabed sediment type and bathymetry on the propagation and received levels of ADDs associated with salmon aquaculture. Potential exposure risks to marine mammals were also investigated, based on published audiogram data and assumptions on animal behaviour. Results indicate that ADDs may be capable of causing hearing damage in marine mammals which remain close (<500 m) to the sound source over periods of several hours. The risk appears to be particularly acute for seals which are often observed foraging around fish farms over such time-scales.

Underwater explosions generate especially high levels of anthropogenic noise. After World Wars I and II at least 500 000 tons of explosive ordnance remained undetonated in the North Sea. On the Dutch Continental Shelf, around 100 controlled detonations of unexploded ordnance are carried out by the Dutch Royal navy annually. Such irregularly occurring events can trigger behavioural responses (i.e. flight) and lead to temporary or permanent hearing loss in marine mammals. Modelling studies suggest that thousands of porpoises in Dutch waters could be affected by hearing loss every year (von Benda-Beckham *et al*., 2015; Aarts *et al*., 2016), highlighting the potentially very large scale of this threat and the need an appropriate risk assessment strategy and mitigation measures (see also Koschinski, 2011; ASCOBANS, 2015). Koschinski and Kock (2015) documented cetacean-friendly alternatives for removal of explosives, including the use of robotic equipment, freezing, Water Abrasive Suspension cutting, disposal in a Static Detonation Chamber and photolytic destruction of explosive substances. In relation to mitigation they proposed the use of bubble curtains around the ordnance during detonation and the deployment of pingers or other acoustic scaring devices prior to and during detonation.

GERMANY: Brandt *et al*. (2016) analysed the effects on harbour porpoises of the construction of eight offshore windfarms within the German North Sea between 2009 and 2013, by combining passive acoustic monitoring (2010–2013) and aerial survey records (2009–2013) of porpoises with data on noise levels, other pile-driving characteristics and noise mitigation methods (if any). Results indicated significant declines in porpoise detections at distances of up to 17 km from the noise source, with noise levels exceeding 143 dB SEL_{05}, comparable to levels identified as significant for harbour porpoises in previous studies (e.g. Lucke *et al*., 2009; Dähne *et al*., 2013; Kastelein *et al*., 2013; Diederichs *et al*., 2014). Effects of pile-driving operations on porpoise distribution were seen only over comparatively short periods (1–2 days), suggesting that negative effects of windfarm construction at the population level were unlikely.

Brandt *et al*. (2016) also reported significant declines in porpoise detections at ranges of up to 10 km prior to pile-driving operations, particularly under low windspeed conditions. It was suggested that noise associated with the pre-construction activity of support vessels could travel further under these conditions, indicating that effects of both support vessels and weather on overall noise propagation should be considered in future impact assessments.

NETHERLANDS: von Benda-Beckham *et al*. (2015) used information about unexploded ordnance from the Netherlands Ministry of Defence and data on the seasonal distribution of porpoises to generate maps of exposure of porpoises to noise from explosions in Dutch waters. They estimated that 88 explosions annually would lead to permanent hearing loss in between 1280 and 5450 harbour porpoises. A follow-up study by Aarts *et al*. (2016) modelled porpoise distribution based on aerial survey data and predicted that, every year,, depending on the assumptions made about porpoise distribution, between 1100 and 1200 harbour porpoises would suffer permanent threshold shifts (PTS) and a further 15 000 to 24 000 porpoises would suffer temporary threshold shifts (TTS).

UK: Russell *et al*. (2016) reported on a telemetry study of tagged harbour seals in the Wash (southern North Sea) adjacent to areas where windfarms were being constructed. During pile-driving operations, seal site usage was significantly reduced up to 25 km away from the pile-driving site. This effect appeared to be temporary: within two hours after pile-driving ceased, seal distribution had returned to pre-pile-driving patterns. There was no evidence that seals were displaced from windfarm construction sites over longer time-scales, or that displacement occurred once windfarms had become operational (see also Russell *et al*., 2014).

Morell *et al*. (2015) described the morphology of the odontocete inner ear based on scanning and transmission electron microscopy, noting that structural alterations potentially attributable to sound exposure can be observed in very fresh stranded cetaceans, provided that the organ of Corti is fixed within 18 hours post-mortem. Morell *et al*. (2017) detected change in the structure of the organ of Corti in a pilot whale from a mass stranding event on the Scottish coast and concluded that this approach could be adopted as a routine protocol to investigate noise overexposure in stranded cetaceans.

Merchant *et al*. (2016) reported on recent efforts to develop a monitoring network for anthropogenic noise in UK waters, to meet MSFD requirements. Considerable regional differences in anthropogenic noise exposure were evident, largely driven by variations in the distribution of shipping and fishing effort. New approaches to improve statistical rigour in proposed noise monitoring programmes were presented and the authors concluded that MSFD indicators, as currently defined, may need to be reconsidered to improve monitoring efficacy.

PORTUGAL (Azores): Romagosa *et al*. (2017) report low-frequency (18–1000 Hz frequency band) underwater noise levels off the Azores, in areas used intensively by baleen whales to forage. Bottom-fixed hydrophones, deployed at three different seamounts, were used to measure background noise levels and ship noise and estimate the percentage of time with noise levels above 120 dB re 1 μPa. Average noise levels at each location varied between 92.9±6.6 to 97.6±8.5 dB re 1 μPa. Contribution of local boat noise to background noise levels varied between 16–19 dB, depending on the study site, and on average was nearly 10 dB higher than the wind contribution. Maximum percentage of time with levels above 120 dB re 1 μPa per month was 3.3%. These results suggest that noise levels in the Azores are generally low and unlikely to cause significant behavioural disturbance to baleen whales. However, the authors stress the importance of monitoring other areas in the archipelago closer to ferry routes, commercial shipping routes or routinely used by whale watching boats, as these areas are expected to be considerably noisier.

USA: The US National Marine Fisheries Service (NMFS), in collaboration with the US National Oceanic and Atmospheric Administration (NOAA), recently published updated technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (NMFS, 2016). This guidance includes updated acoustic thresholds for the onset of temporary and permanent threshold shifts (TTS and PTS respectively) in marine mammal hearing following acute, incidental exposure to underwater anthropogenic sound sources. This guidance follows the same general approach as originally advocated by Southall *et al*. (2007) but incorporates subsequent advances in knowledge. One important advance is the explicit consideration of exposure duration through the definition of a novel metric (Cumulative Sound Exposure Level, SEL_{cum}) that accounts for accumulated exposure to a single activity/sound source over time.

2.3.5 Ship strikes

Mortality due to ship strike has become a globally important conservation issue for cetaceans (Fais *et al*., 2016). The IWC's Conservation and Scientific Committees are both working on ship strikes and IWC also hosts a ship strike database. Evidence of ship strikes comes from a range of sources among which direct observations from vessels and examination of dead animals are likely the biggest. Areas with high densities of whales and ships unsurprisingly tend to have high reported mortality rates due to collisions but certain species seem to be more prone to ship strikes (Laist *et al*., 2001; Fais *et al*., 2016). The largest number of ship strikes has been reported for fin whales, but humpback, sperm and grey whales are also commonly hit by ships (Laist *et al*., 2001). Population level effects are most likely for small populations and concerns have been raised for North Atlantic Right Whales (Kraus *et al*., 2005). Ship strike mortality in smaller odontocetes seems to be relatively infrequent, e.g. causing 1–4% of harbour porpoise mortalities (Evans *et al*., 2011; Jepson *et al*., 2016).

2.3.6 Cumulative impacts

NETHERLANDS: An approach was tested to model the cumulative effects of the construction of offshore windfarms and seismic surveys on harbour porpoises. The number of animal disturbance days, as derived from noise emissions and noise propagation models, was used as an indicator of the population changes that were predicted with the Interim PCoD model. In the worst case scenario describing the international activities in Dutch North Sea waters as well as surrounding countries, for seismic surveys a reduction of 53 000 porpoises in the North Sea was modelled, the scenario for windfarms predicted a population reduction of 46 000 animals. The caveats of using this approach for assessing the impact on the North Sea harbour porpoise population are discussed in detail in Heinis *et al*. (2015).

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3 Tor B. Review and update the criteria for assessment of cetaceans in the context of the MSFD

In 2014 ICES provided initial advice on possible cetacean indicator targets, as follows:

> "M-4aA suitable indicator target for harbour porpoises could be 'For each assessment unit, maintain harbour porpoise population size at or above baseline levels, with no decrease of ≥30% over a three-generation period (36 or 22.5 years).' The 22.5 years are based on data primarily from OSPAR Regions II and III and is therefore preferable.
>
> M-4b A suitable indicator for inshore bottlenose dolphin could be 'For each assessment unit, maintain inshore bottlenose dolphin population sizes at or above baseline levels, with no decrease of ≥30% over any ten-year period.'
>
> M-4c A suitable indicator for offshore bottlenose dolphin could be 'Maintain the offshore NE Atlantic bottlenose dolphin population size at or above the baseline level, with no decrease of ≥30% over a three-generation period (63 years).'
>
> M-4d A suitable indicator target for white-beaked dolphins could be 'Maintain the white-beaked dolphin population size at or above the baseline levels, with no decrease of ≥30% over a three-generation period (54 years).'
>
> M-4e A suitable indicator target for minke whales could be 'Maintain the minke whale population size at or above the baseline levels, with no decrease of ≥30% over a three-generation period (66 years).´
>
> M-4f A suitable indicator target for common dolphin 'Maintain the Northeast Atlantic common dolphin population size at or above the baseline level, with no decrease of ≥30% over a three-generation period (44 years).'" (ICES, 2014a).

This was largely reiterated in the 2016 Advice to OSPAR (ICES, 2016a). It was also noted that the information available was insufficient to make assessments against the proposed indicator targets, for all species except harbour porpoise (ICES, 2016a). However, the indicator does not cover all eventualities and as such requires refinement. ToR B was therefore developed, specifically noting that there is a need to revise the criteria for determination of whether abundance for an Assessment Unit has fallen below the baseline level and to review the decision to adapt the IUCN Red List criteria for identifying vulnerable species.

The IUCN Red List criteria define a species as 'vulnerable' when 'an observed, estimated, inferred, projected or suspected population size reduction of ≥30% over any ten year or three generation period, whichever is longer (up to a maximum of 100 years in the future)". However, this approach does not, for example, indicate whether a species that has declined by 40% in four generations would be considered vulnerable, nor does it identify a baseline. The rationale for the lack of reference to a baseline in the IUCN criteria is that, since the environment is continually changing, how a population relates to an explicitly defined point in the past is not considered useful. For MSFD purposes, however, a baseline should be defined. This baseline could for example be the start of the available time-series of data, independently of whether the time-series extends across three, five or ten generation times. IUCN does not define the time period over which the trend should be measured and there is no requirement to wait for ten years or three generations before calculating the trend. In the context of the MSFD, it may be useful to define a time-scale. In the case of oceanic cetaceans, the minimum period over which abundance trends could be assessed would be the interval between consecutive large-scale surveys (currently eleven years).

Although the baseline should, in principle, be identified based on good historical data on abundance at an appropriate spatial scale, such data are not available for any cetacean species in the MSFD area, although some indirect and local-scale evidence exists about historical trends. Historical abundance and distribution are therefore essentially unknown. In addition, some current populations of cetaceans are of recent origin. For example, there is no evidence that there were bottlenose dolphins in the Moray Firth 100+ years ago. Even in cases where numbers are suspected to have declined, the populations could probably not realistically be restored to historical levels because today's marine environment is very different, in part due to climate change and human impacts. Consequently, a recent baseline must be utilized, which can however then be assessed as either representing a "normal" situation (consistent with Good Environmental Status), or one that is degraded. The most useful baselines for abundance of wide-ranging cetacean species in European Atlantic waters derive from the results of large-scale dedicated surveys such as SCANS and CODA (e.g. CODA, 2009; Hammond *et al*., 2002; 2013; 2017).

Cetaceans are widely distributed in a range of marine habitats and are abundant throughout the OSPAR Regions II, III and IV. Dedicated large-scale cetacean surveys in this area currently occur approximately once every decade (e.g. SCANS). These purpose-designed aerial and/or shipboard surveys, using line-transect distance sampling methods, offer two methods to estimate abundance, namely design-based and model-based (Buckland *et al*., 2001; Hammond *et al*., 2013). The latter estimates come from models fitted to the survey data to generate a density surface from which abundance can be derived (e.g. Gilles *et al*., 2016; Rogan *et al*., in press).

For most cetacean species in the part of the NE Atlantic covered by the MSFD (i.e. the EEZs of EU member states and adjacent oceanic waters), only two large-scale abundance estimates are currently available (i.e. SCANS II July 2005 (along with CODA in 2007, which covered deeper waters) and SCANS III July 2016), so no assessment of a trend involving change from a baseline (or a certain rate of decline) is statistically feasible; although in principle the statistical significance of the difference between the two estimates could be assessed. For harbour porpoise, white-beaked dolphin and minke whale in the Greater North Sea there are three design-based estimates (SCANS I July 1994, SCANS II July 2005 and SCANS III July 2016), so an assessment of the trend is possible, although the statistical power to detect change was relatively low (ICES, 2014b; see also ToR e for results of new power analyses following the 2016 survey), i.e. only a high rate of decline would be statistically significant. Clearly, the situation could be improved by increasing the frequency of such surveys, for example in line with reporting related to the MSFD.

Two other versions of the indicator, distinct from the IUCN criteria, were also considered:

1) Adoption of similar approach to that used for defining thresholds and trends in the Favourable Conservation Status reporting under the Habitats Directive. A species is considered favourable in terms of its population if abundance is at or above the favourable reference population (FRP). As a minimum requirement, the FRP equates to the size of the population within a Member State's territory in 1992 when the legislation was enacted. One determinant of an 'unfavourable' condition for a population is an abundance estimate 25% or more below the FRP. With regard to trends, the European Commission (2011) states:

   'The reporting period for the Habitats Directive is six years but estimates of trend are more likely to be statistically robust over longer time periods. It is therefore recommended to estimate [short-term] trend over two reporting cycles, i.e. 12 years (or a period as close to this as possible), as this should give a more reliable and comparable estimate of the trend. Long-term trends, which are likely to be more statistically robust, can also be reported (in a series of optional fields). The recommended period for assessing longer term trends is four reporting cycles (24 years).'

2) Adoption of criteria proposed at the national level, as exemplified by the UK. The UK abundance indicator is defined as 'At the scale of the MSFD subregions, abundance of cetaceans is not decreasing as a result of human activity: in all of the indicators monitored, there should be no statistically significant decrease in abundance of marine mammals caused by human activities.' The target is thus: 'no statistically significant decrease in abundance as a result of anthropogenic activity.' The baseline utilised for the assessment is likely to be SCANS I (North Sea) and SCANS II (Celtic Seas or for European waters as a whole).

Both these approaches refer absolute declines rather than rates of decline, and use a specific baseline against which to determine whether the change in the population size has been positive, stable or negative. The first option makes a recommendation about the time period to be considered. However, there are issues with both options. The first takes no account of differences in the biology of different species. The second requires determination of the cause of any negative trend, which may be difficult (as exemplified by the work undertaken in the UK to determine the causes of declines in harbour seals in some colonies (e.g. SCOS, 2014; 2015; 2016; SMRU, 2012; Marine Scotland, 2016).

Given the long life cycle of many cetaceans, WGMME considers framing the indicator within the IUCN context to be biologically appropriate but agrees that the wording of the indicator should be improved, to include consideration of absolute amounts of decline relative to a baseline.

It is proposed that the indicator be rephrased as:

> For each assessment unit, maintain [insert species name] population size at or above baseline levels (using the earliest reliable population estimate (e.g. from SCANS I or II) as the baseline), with no absolute decrease of >30% and a rate of decrease no greater than 30% over three generations. The rate of decrease may be assessed over a shorter time period if the projection of future decline is consid-

ered to be reliable and/or the absolute decline is so large that the population is considered to be at risk.

An alternative would be to specify no significant decline below the baseline. However, it is unrealistic to expect that cetacean populations would show no natural fluctuation in size. In addition, even for large-scale dedicated sightings surveys, the CV is unlikely to be less than 0.15 and it is not certain that a 25% decline (as used in the Habitats Directive) would be detectable, since it depends on survey periodicity and the length of the time-series (see Tables 2.1 and 2.2). The selection of 30% as a threshold is therefore both pragmatic and consistent with the IUCN criteria. In any case, it is not necessary to wait for a 30% decline to occur. A reliable projection of such a decline is sufficient to trigger action.

Recommendation to ICES, the EU, Member States and associated states: Our ability to detect declines in cetacean abundance remains somewhat limited, and this situation could be improved by increasing the frequency of large-scale surveys to once every six years, as favoured by OSPAR. Without large-scale surveys like the SCANS surveys, no assessments of cetacean abundance in European Atlantic shelf waters would have been possible. WGMME therefore recommends that large-scale surveys of cetaceans in European Atlantic shelf waters should be brought into baseline monitoring, coordinated between all relevant coastal states, and repeated every six years. The annual cost per nation would be modest and the current situation, whereby surveys are funded on an *ad hoc* basis, is risky.

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4 ToR C. Review current issues in relation to direct impacts of seals on fisheries

ToR c aims to address current issues in direct (operational) seal–fisheries interactions in the context of (in some cases increasing) seal population abundance and distribution. In some areas there is increasing concern from fisheries organisations regarding the survival of coastal passive gear fisheries and pressure for economic compensation and/or targeted removal of 'rogue' seals. Another major aspect of seal–fisheries interactions includes competition for shared resources; however, in 2017 the group aims to review only direct interactions such as depredation. Indirect interactions (e.g. competition for food and transmission of codworm and other parasites) could be reviewed in 2018.

4.1 Introduction

Interactions between fisheries and seals can have many facets. Effects can be direct as well as indirect and can influence either or both sides negatively. The present review aims to provide a broad overview on current issues regarding direct effects of seal depredation on the fishing and aquaculture industries within the ICES area, also drawing on examples from the Mediterranean. Coram *et al*. (2016) defined depredation as "an act of predation where the prey is already held captive, either by being caught in wild capture fisheries equipment or as part of an aquaculture system". Hence, the review will cover seal-induced damage to the fishing and aquaculture industries, considering the extent of damage to the catch as well as estimation of economic losses due to depredation. Further, we explore experiences with mitigation measures. The review focuses mainly on recent knowledge, from 2012 to the present day. It should be noted that this is not an exhaustive literature review, rather a synthesis of information collated by WG members. Further, anecdotal information was not taken into consideration here. In Table 4.1 we have summarized recent information on the fishery target species of concern, the proportion of the total catch that is damaged due to seal depredation, and estimated costs due to depredation, as well as mitigation methods applied in different areas within the ICES region.

Most of the information presented here concerns grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*), also ringed seals (*Pusa hispida*) in the Baltic Sea and monk seals (*Monachus monachus*) in the Mediterranean Sea. However, it is likely that almost all pinniped species present in the area are involved in depredation, including hooded seals (*Pagophilus groenlandicus*), harp seals (*Pagophilus groenlandicus*) and bearded seal (*Erignathus barbatus*). Only the walrus (*Odobenus rosmarus*), which feeds mainly on benthic molluscs, is unlikely to forage at fishing gear.

Clearly seal-fishery interactions are not a new phenomenon in the ICES area and there is an extensive literature on depredation, bycatch and mitigation measures (see, for example, Northridge, 1984; 1991; Würsig and Gailey, 2002; Bosetti and Pearce, 2003; Lunneryd *et al*., 2003; Lehtonen and Suuronen, 2004; Fjälling *et al*., 2006; 2007; Suuronen *et al*., 2006; Westerberg *et al*., 2006; Andersen *et al*., 2007; Königson *et al*., 2007a,b; 2009; Hemmingsson *et al*., 2008; North Eastern Sea Fisheries Committee, 2008; Götz *et al*., 2010; Butler, 2011; Graham *et al*., 2011; Königson, 2011; Varjopuro, 2011).

In some areas, depredation is increasing in parallel with increasing seal populations. New locations and new types of fisheries have become targets for seal depredation. As an example, all three seal populations that breed in Swedish waters (harbour seals, ringed seals and grey seals) are increasing and a large increase in depredation has been observed. Depredation on cod fisheries in the southern part of the Swedish east coast was previously not a large problem, but has recently increased significantly. As a result, new fishing gear is being developed to mitigate the depredation problem. In Sweden, the proportion of damaged fishing gear increased from 13% to 32% between 2010 and 2014 for gillnet fisheries and from 13% to 49% for longline fishing during the same period (Havs- och vattenmyndigheten, 2014).

4.2 Seal species of concern and "rogue seals"

A range of seal species is involved in depredation events affecting fisheries and aquaculture throughout the ICES area (Table 4.1). For example, Northridge *et al*. (2013) found that, in Scotland, both grey seals and harbour seals depredate on salmon farms and in the southern part of the ICES area monk seals are responsible for depredation from several different types of fishing activity including gillnet fishing and longlining (Ríos *et al*., 2017).

It is worth noting that concern about grey seal depredation on salmon in fixed nets along the Scottish coast can be traced back at least 60 years (e.g. Rae, 1960; Rae and Shearer, 1965), thus pre-dating the arrival of salmon aquaculture in the early 1970s. This led to frequent calls for control of the grey seal population and seals were routinely shot near salmon netting stations. Monk seal fishery interactions also have a long history; for example, fishery bycatch and deliberate killing, likely by fishermen, were among the main identified causes of death of monk seals in a study on their diet in the eastern Mediterranean in the mid-2000s (Pierce *et al*., 2011).

A few studies have investigated effects of factors such as sex and age on depredation behaviour. Kauhala *et al*. (2015) compared the age, sex and body condition between Baltic grey seals shot around fishing nets and grey seals that were bycaught. They found that most bycaught seals in spring were pups, with significantly thinner blubber layer, while those bycaught in autumn were subadult and adult males, in good condition. Animals that were shot around fishing nets were mainly adult males in good conditions. Thus, it was concluded that "rogue" seals were not a random sample of the population and it was suggested that sexual dimorphism could partly explain this bias (larger males needing more energy, therefore engaging in a more profitable, albeit dangerous, foraging strategy such as depredation).

Harris *et al*. (2012) suggested that net specialist grey seals are responsible for depredation at salmon farms in the Moray Firth (Scotland). Few seals were involved in depredation, with most individuals observed around nets coming back each year. These results are supported by Harris *et al*. (2014) who photo-identified grey seals around a salmon farm in Scotland, and found that two individuals made 63% of the seal visits to the farm. Diet analysis of grey and harbour seals bycaught or shot around salmon nets however showed that these seals did not feed exclusively on salmonids (as indeed was apparent from analysis of stomach contents of seals shot near coastal salmon nets in Scotland in the 1950s; Rae, 1960). Harris *et al*. (2012) suggest that bycaught seals may usually be naïve seals, while specialist adult seals, which are probably responsible for most fish depredation, more seldom get caught in the nets. Königson *et al*. (2013) found that adult male grey seals specialize in depredation in the Baltic salmon trap fishery and that eleven photo-identified grey seals made 71% of the 600 recorded visits. Research in Finland also showed that male grey seals which were caught in salmon traps and subsequently equipped with satellite tags repeatedly visited these traps, supporting the theory of single specialised seals as so called "rogue" seals (Lehtonen *et al*., 2013).

4.3 Consequences of depredation

When seals attack fishing gear or aquaculture equipment, fish can be damaged or completely removed. Another consequence of seal depredation is physical damage of fishing gear and aquaculture pens leading to costs associated with mending the gear. Further, damaged equipment can enable fish to escape. Holes caused by seals are the single most frequent cause of salmon escape from aquaculture farm cages (Northridge *et al*., 2013). Within the fish-farm industry, growth reduction of stock, as well as fish welfare concerns, have been reported as a consequence of seal depredation (Coram *et al*., 2014).

The economic loss due to seal depredation is estimated to be large in some areas but it is often difficult to estimate. A study in Sweden tried to estimate the "hidden loss" and concluded that an estimated 15–36 % of the total potential catch were lost to seal depredation (Köningson *et al*., 2009). In salmon farms in Scotland in 2011, seals were held responsible for the escape of around 21 000 fishes. Although this figure is most likely underestimated, these large-scale escape events are regarded as commercially less significant in contrast to the almost daily losses due to direct depredation in some fishfarms. Over a study period of ten years, in which 87 fish farms were monitored, an estimated 1.4 million salmon deaths were attributed to seal predation resulting in an estimated annual loss of GBP 2.5 million if those salmon had reached the expected size of 5 kg each (Northridge *et al*., 2013). Another example, where seal depredation is significantly affecting the industry is the salmon and cod fisheries around Bornholm, Denmark. In some cases, over 20% of the total daily catch was damaged. There is however large annual variation and depredation also varies between areas, which should be taken into consideration when the amount of loss is estimated (Larsen *et al*., 2015).

Table 4.1. An overview of current and known information on seal and target species of concern for seal depredation, the amount/proportion of total catch that is damaged due to seal depredation, estimated costs due to depredation as well as mitigation methods for different.

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
Grey seals	Bluefish, different squid species (especially *Doryteuthis pealeii*)	Fish weirs	-	Nantucket Sound, USA	NA	-	-	(Nichols *et al.*, 2014b)	Sonar and video was used to assess depredation. Seals mainly depredated during night. Seals also caused potential catch to exit the gear
Grey seals	Longfin inshore squid, Atlantic menhaden, sea robins, black sea bass	Fish weirs	Loss of total catch: Summer flounder (*Paralichthys dentatus*) (5%), *C. striata* (3%), *D. pealeii* (2%) and *Prionotus* spp. (2%).						
Total weight of partially consumed catch was <1% of total catch weight.	Nantucket Sound, USA	NA	-	-	(Nichols *et al.*, 2014a)	Amount most likely underestimated, method is based on the estimated weight of remnants and does not account for fish totally swallowed or schools being disturbed and thus			

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
						not entering the trap			
Monk seals	Parrot fish, dusky spinefoot, saddled sea bream, red scorpionfish, common cuttlefish, Common **dentex**, common pandora, garfish and common octopus	Gillnet, trammelnet, gill/ trammelnet, longline	Depredation in 19.1% of fishing trips; damage to nets and damage to catch (obvious bite marks) occurred simultaneously on 67 occasions (44.0% of fishing trips)	Lipsi Island, Greece	Aegean-Levantine Sea	2230€ /fishermen/year catch loss	-	(Ríos *et al.*, 2017)	The greatest probability of damage was observed on gill nets; greater probability of damage by monk seals in spring and summer; the catch per unit of effort of four species (*S. officinalis*, *S.cretense, P. erythrinus*, *O. melanura*) was lower when there was monk seal interaction than when there was not.
Grey and harbour seals	Salmon Farming	Mainly cages	Average of 264 lost salmon per stocked month --> data from 87 fish farms over ten years	Scotland	Celtic Seas	estimated annual loss of GBP28.736/farm site --> data from 87 fishfarms over ten years	ADD, predator nets, hunt (e.g. in 2011 242 seals were shot under licence (wider western	(Northridge *et al.*, 2013)	

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
							highland region: 58 harbour seals, 37 Grey seals)), regular removal of dead fish from the net bottom		

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
Grey seals	Lobster, cod and groundfish, skates, dogfish, and monkfish	Gillnets, handlines, lobster pots	Gillnet fisheries: depredation occurred on 39±28% of commercial trips (N = 20)	Cape Cod, USA	NA	Estimated at $1,887,940 USD by a total of 73 respondents (cf. comments)	-	(Gruber, 2014)	Lost time and effort = 60% of financial losses due to grey seals, depredation = approximately 29% Rest = costs of gear repair/replacement, extra fuel, and catch affected by seal worm. Detailed costs per fishery and damage type see pp. 26–30.
Grey and harbour seals	Primarily anglerfish; hake, turbot	Offshore passive (e.g. gill, tangle)		Celtic and Irish Seas	Celtic Seas		-	(Cosgrove *et al.*, 2015a)	
Grey seals		Longline		Faroe Islands	Faroes			(Werner *et al.*, 2015)	
Grey, ringed, and harbour seals	Cod, salmonids, whitefish, vendace, pike, pike perch, eel, bream, flat-	Passive gear; Trapnets, hooks, nets		Swedish coast	Greater North Sea; Baltic Sea	33 300 000 SEK in 2014	Experiments with cod pots. Economical compensation 16 m SEK in 2014. Protective hunt of	(Havs- och vattenmyndigheten, 2014)	

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
	fishes, herring, lumpsucker, mackerel, wrasse, etc.						seals		
Grey and harbour seals	Pollock, hake and monkfish	Set-net fisheries	Pollock= 18%, hake=10%, monkfish=59% of landings were depredated by seals (Total loss of landing could rise to over 50% when estimate of fish that are totally removed are taken into account)	Irish waters	Celtic Seas	Total annual value of seal damaged fish for pollock and hake is 1.7mil Euro (Direct effect of seal predation on catches)		(Cosgrove *et al.*, 2013)	
Grey and harbour seals	Cod, salmonids, lumpsucker & flatfish	Passive fishing gear (nets, longline, trapnets)	0–25% of total catch had seal injuries (Jutland west coast; 6–25% of the total cod catch, Bornholm; 0–20% salmon	Danish waters	Greater North Sea; Baltic Sea	5–25 000 DKR/month/boat (cod fisheries on Jutland west coast; 5–25 000 DKR/month/boat, Lumpsucker fisheries in Djursland	Experiments with developing seal safe fishing gear, together with Sweden	(Larsen *et al.*, 2015)	

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
			and cod fisheries, North of Djursland; 7% of Lumpsucker catch).			7000 DKR/month/boat).			

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
Harbour seals	Cod, Haddock, Winter skate, Yellow tail, flounder, Spiny dogfish, pollock, Monkfish	Gillnet	0.4% caught discarded due to seals in total	Georges Bank, Massachusetts, USA	NA	Loss due to depredation estimated to 0.7% of the market value ($435.96 out of the estimated market value of catch during the study which was $61 792.55)		(Rafferty *et al.*, 2012)	
Grey and harbour seals	Primarily pollock, anglerfish, cod, hake, ling, anglerfish, hake, turbot	Inshore passive (e.g. gill, tangle, driftnets) and offshore passive (e.g. gill, tangle)	20–30%	Celtic and Irish Seas	Celtic Seas		Cessation of fishing activities, frequent relocation of gear, avoidance if	(Cronin *et al.*, 2014)	No quantitative information available; qualitative information solicited from fisheries representatives (n = 6)

Seal species	Fish/prey species	Fishing gear type	Estimated loss of catch due to seal predation	Specific area	ICES ecoregion	Estimated cost	Mitigation	Reference	Comments
							seal dense areas		

4.4 Mitigation

One solution to wildlife–human conflicts is developing non-lethal mitigation techniques (Blackwell *et al*., 2016). Over the years several mitigation measures aimed at reducing seal depredation and gear damage have been tested. These include modification of fishing gear and technique, and the use of Acoustic Deterrent Devices (ADDs) in order to reduce the presence of seals in the respective area, as well as the targeted removal (shooting) of seals around fishing gear.

4.4.1 Gear and method optimisation

Depredation in Irish bottom-set gillnets has increased in the last few years. Cosgrove *et al*. (2015b) suggests that, in inshore gillnet fisheries for pollock species (*Pollachius* spp.) in shallow waters, soak time should be kept short to reduce seal depredation. However, for the deeper, more offshore, gillnet fishery for hake (*Merluccius merluccius*), reduction of soak time did not reduce depredation rates and the authors therefore suggested that, in deeper waters, systems actively deterring seals should be tested.

Codpots are regarded as being relatively safe from seal depredation and are seen by some as a viable alternative to gillnets. However, as the catch efficiency is not as high as for gillnets, the design needs to be optimised to increase rates of acceptance and use by fishers (Hedgärde *et al*., 2016; Ljungberg *et al*., 2016; Stavenow *et al*., 2016).

As yet there is no one solution which satisfies all needs with regard to catch efficiency, practicability and protection from seals and further research to improve methods is necessary. Stavenow *et al*. (2016) found that pots designed with loose netting around the upper chamber (in contrast to tightly stretched mesh) attracted more seals and received most attacks. However, mesh size and material were not correlated with seal presence or attack behaviour (Stavenow *et al*., 2016). Königson *et al*. (2015) compared different shapes and sizes of Seal Exclusion Devices (SED), using vertically mounted metal frames adapted to the fish pots. Although designed to reduce seal bycatch, the modified gear also reduced depredation by seals. The study showed that the use of SEDs had no catching power of the pots but reduced seal bycatch to zero.

In fish farms, net tensioning is believed to be a key factor in decreasing depredation by seals. Coram *et al*. (2014) showed that seals can utilise loose nets to create pockets in order to trap and consequently catch fish. Coram *et al*. (2016) used underwater cameras to observe harbour and grey seal attacks on salmon pens. To understand the physical force that individual seals can exert on the net when trying to reach the fish, seals were purposely trained to push against the net. Motivation and experience turned out to be key factors determining the force seals are able to generate while pushing the net. Individual differences in learning speed is probably also a factor to consider (Coram *et al*., 2016). Stavenow *et al*. (2016) also documented individual differences in learning ability and attack behaviour amongst seals that should be taken into consideration when managing seal depredation issues.

Another method of depredation mitigation is the installation of anti-predator nets which are placed around the actual fish trap or farm cage in order to prevent seals from accessing the inner nets. However, this method has the potential to cause several problems, such as increased rates of entanglement of marine wildlife, problems with deployment and holding the construction in place, as well as increased problems with the maintenance and operation of the actual net or cage through reduced water exchange, fouling and increased drag. The use of stronger netting material has reduced the damage to nets by seals but has led to an increased rate of fish being taken and consumed through the net and thus has not necessarily reduced the rate of depredation. Finally, removing dead fish from the bottom of the net or cage has reduced rates of depredation as has the installation of a second bottom to reduce the likelihood of seals sighting the prey (Northridge *et al*., 2013).

4.4.2 ADDs

Currently, acoustic deterrent devices (ADDs) are widely used to deter marine mammals for several reasons, such as displacing them from areas where injurious levels of noise pollution are expected (e.g. windfarm construction sites) but also to protect fish farms and fishing equipment (Coram *et al*., 2014). In a review of information available about seal-fisheries interactions in Irish waters, Cronin *et al*. (2014) reported that seal ADDs are in use at many salmon farms in Ireland, but evidence suggests that seals become habituated to the noise and ADDs are not effective in the long term. Hence, the timing of use of ADDs is a key factor. In Scotland, seal depredation rates on aquaculture farms increased over the first six to seven months of the production cycle, with the greatest depredation intensity around the months nine and ten (Northridge *et al*., 2013). Delaying the implementation of mitigation measures until towards the end of the production cycle, when the seal depredation peaks, could help minimise the likelihood of seal habituation towards the ADD.

Götz and Janik (2013) reviewed the efficacy of, and concerns about, using ADDs to prevent pinniped predation at fish farm. They found that ADD efficacy is highly variable in wild deployments, in terms of both the magnitude and duration of effects. Possible reasons for this variation in efficacy include differences in the ADD deployment method, the variable foraging motivation of the target animals, differences in sound propagation in different areas, and between-population and between-species differences in natural reactions to sound.

Despite obvious positive effects of using ADDs, some concerns have been expressed about negative consequences. Due to the widespread use of ADDs for deterring pinnipeds around fish farms in Scotland, an area as large as 15% of Scottish inshore waters could be affected (Coram *et al*., 2014).

Concerns about the potential to cause hearing loss or impairment in the target and non-target species were reviewed by Götz and Janik (2013). Exclusion of odontocetes from areas where ADDs are deployed is well-documented; the authors suggest that this is due to their sensitivity to the devices' frequency range. They also suggest that the use of lower frequencies may mitigate this effect but note that lower frequencies could have greater impacts on baleen whales and some fish. They propose that devices which elicit autonomous reflexes related to flight behaviour (e.g. the startle reflex) might be more effective than the devices available at the time.

The use of ADDs in marine salmon farms was tested in western Scotland during a 19-month experiment (Götz and Janik, 2015; 2016). Visual monitoring indicated that the numbers of seals within 100 m of the nets were only slightly lower during the experimental deployments, but the use of ADDs reduced the loss of fish by 91%. In contrast to other studies, this study showed that harbour porpoises and otters were not affected by the acoustic devices due to the specific adaption of the deterrent signal to take account of the hearing frequency bands of target and non-target species (Götz and Janik, 2016). This way of adapting such devices is seen as an important step in reducing adverse effects on non-target species. Trites and Spitz (2016) commented on the study by Götz and Janik (2016) and suggested that further investigation with larger samples sizes and in other areas are necessary to test for consistency of the results.

4.4.3 Lethal removal of seals

One method of reducing seal-fisheries interaction is killing of seals, either by targeted removal (under licence) of "rogue" individuals that might have specialised on depredation from nets or aquaculture pens (protective hunt) or by licensed hunt with the aim of reducing the total number of seals in a population (cull). Information on seal hunting statistics is usually scarce in the literature and the extent of lethal removal as a response to seal fisheries interactions is therefore difficult to assess quantitatively. However, as an example, 242 seals were shot under licence in 2011 in Scotland around fish farms, with 58 harbour and 37 grey seals reported being shot in the wider western Highland region (Northridge *et al*., 2013). Protective hunting was also applied in Swedish waters, from 1967 until 1988 and again from 2001 until today. In 2015 the Swedish government allowed a protective hunt of a maximum of 350 grey seals and 160 harbour seals (Naturvårdsverket, 2015). Hunting of grey seals is regulated in Sweden in relation to their population development and abundance, so as to prevent excessive mortality. In general, the effectiveness of culling is highly controversial and there is little or no verifiable evidence that large-scale lethal removal of marine mammals will reduce economic losses in the fishing industry unless a detrimental percentage of the population (> 50%) is removed and numbers are kept at this low level (Bowen and Lidgard, 2013; Morissette *et al*., 2012). Despite the constant targeted removal of seals in Sweden, depredation rates increased from 13 to 32% for gillnets and from 13 to 49% for longline fisheries between 2010 and 2014 (Havs- och vattenmyndigheten, 2014).

4.4.4 Other

Responses from fishermen indicate that other practical measures taken include relocation of fishing gear more frequently or after depredation events, avoiding areas of high seal abundance (e.g. near haulouts or colonies), and abandoning areas of high seal depredation (Cronin *et al*., 2014).

A short overview by Coram *et al*. (2014) found that other techniques which have been tested, with mixed results, include pursuits with boats, explosives (unspecific and broad adverse effects on other wildlife), playback vocalisation of predators (only short-term effect), electric fields, trapping of seals for subsequent translocation and Conditioned Taste Aversion (CTA) whereby poisoned or tainted food is used, with an emetic for instance (substance that makes the animal vomit, and remember/avoiding that food type later). No CTA trial has been conducted with phocid seals yet.

4.4.5 Conclusion

Further development of solutions to protect fish farms and fishing gear is still needed and it is likely that there is no one solution that fits all, as mentioned by Trites and Spitz (2016). Problems with several of the mitigation methods used to reduce depredation, which have been discussed in this section, are that animals habituate to prolonged stimuli such as ADDs over time. Often, the reward for the seals (an easily accessible prey resource being "served to them" in the nets) is greater than the effort necessary to obtain the reward (Schakner and Blumstein, 2013). Actions preventing seals from seeing the prey and/or preventing the access to it might be more viable in the longer term due to the often reported problem with habituation of seals to ADDs (Trites and Spitz, 2016).

Seal depredation is not the only reason for target fish mortality and hence, to put depredation into a broader context, future research should investigate other mortality risks for target species after capture in the fishing gear. Industrial production procedures in aquaculture, discard of non-target species and depredation due to other fish species are factors that need to be addressed when assessing the significance of seal depredation.

4.5 Direct impact of seals on fisheries–a case study from Latvia in 2016

Approximately five years ago, seals became the main problem facing Latvian coastal fisheries, due to the damage caused to fishing gear and reduced catches. The coastal fishery that is most affected by seals is a mixed fishery using mainly stationary gear. It operates in the coastal waters up to 20 m depth or not more than 2 nautical miles from shore. Gear damage and catch losses due to depredation have substantial economic consequences; fishermen have to abandon the fishery at particular times or even move from coastal areas to river mouths, as is the case near Riga. It was recorded that seals also follow the redistribution pattern of the fisheries. Additionally, there have been severe difficulties in catching the breeding stock of Atlantic salmon for artificial reproduction in recent years. As a result, stress and conflict has intensified between stakeholders, e.g. fishermen and nature conservation specialists. Currently, appropriate solutions to the conflict are limited due to insufficient national legislation. However, compensation mechanisms for financial losses due to depredation are one development under consideration (Plikšs, pers.comm.).

In the central part of the Baltic Sea, two seal species occur regularly: grey seal (*Halichoerus grypus*) and ringed seal (*Pusa hispida*). There are no seal haulouts or breeding areas for either species on the Latvian coast but coastal waters as well as the central part of the Gulf of Riga are used for foraging by the seals. The grey seal population in the Baltic has almost tripled in the past ten years (Härkönen *et al*., 2013). In contrast, the population of ringed seals in the Gulf of Riga has shown a slight tendency to decrease or stay at the present low level (HELCOM, 2016). Evidence from telemetry indicates that ringed seals do not reach coastal areas where fishing takes place, but are more concentrated in the central part of the gulf (Jüssi, pers.comm.).

In order to evaluate the possible effects of damage to fisheries caused by seals in the coastal zone, a pilot questionnaire-based study was conducted in 2016. Questionnaires were distributed to all 141 commercial fishermen units operating in the coastal fishery. Preliminary results were as follows:

1) Replies were received from only 26 fishermen units, mainly in the 1st quarter of the year. Numbers of replies decreased over the following quarters each year and only three units provided replies for three quarters continuously;

2) For fisheries conducted in the coastal zone the damage mainly affects gillnets and trapnets;

3) Representativeness of replies was low: about 5% of gillnet fishing and 14% of trapnet fishing were sampled;

4) Damage to gears and catch losses varied by local counties and by fishing season;

5) The overall preliminary estimate of gillnet damage was around 63 200 EUR (Figure 4.1). This was obtained by extrapolating from reported fishing events with gear damage to the total number of fishing actions from national logbooks and assuming that ~25% of catches are damaged;

6) Catch losses mainly concerned commercially important local fish species: salmon (*Salmo salar*), sea trout (*Salmo trutta*), herring (*Clupea harengus*), smelt (*Osmerus eperlanus*), cod (*Gadus morhua*), perch (*Perca fluviatilis*), pike perch (*Sander* spp.) and vimba bream (*Vimba vimba*);

7) The bycatch of seals reported by questionnaires was 55. That is considerably below the number of dead seals washed ashore (240 seals in 2016, which constituted around 10% of the grey seal population in the Gulf of Riga) (Information from Nature Conservation Agency, Latvia). In 2015 an estimated 208 seals were found dead, fewer than in 2016 (Figure 4.2).

4.5.1 Summary

1) The information obtained so far does not allow a complete evaluation of losses to the fishery but reveal the priorities to be considered.

2) Based on this pilot study, a new questionnaire was developed for 2017 that will be targeted at a subsample of fishermen (by agreement with Institute of Food safety, Animal health and Environment "BIOR"), allowing the information collected on seal damage to be related to logbook statistics.

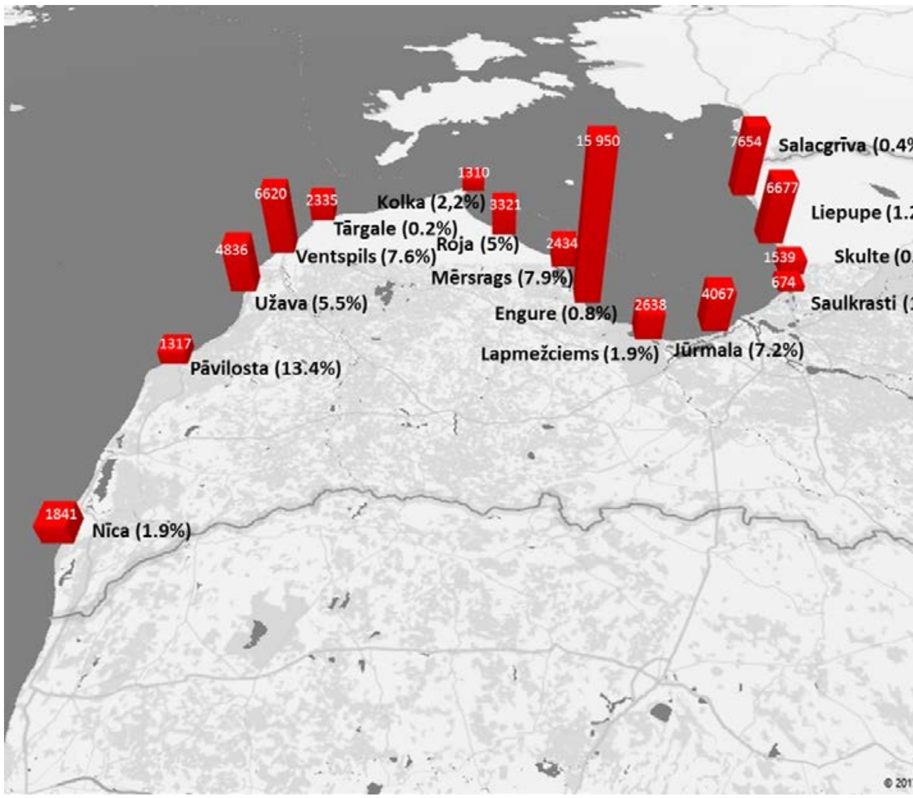

Figure 4.1. Provisionally estimated losses (in EUR) of gillnet damages by seals. Percentage in brackets after local county name represents the share of reported cases of gear damage from questionnaires to the total number of records of gillnet fishery from national.

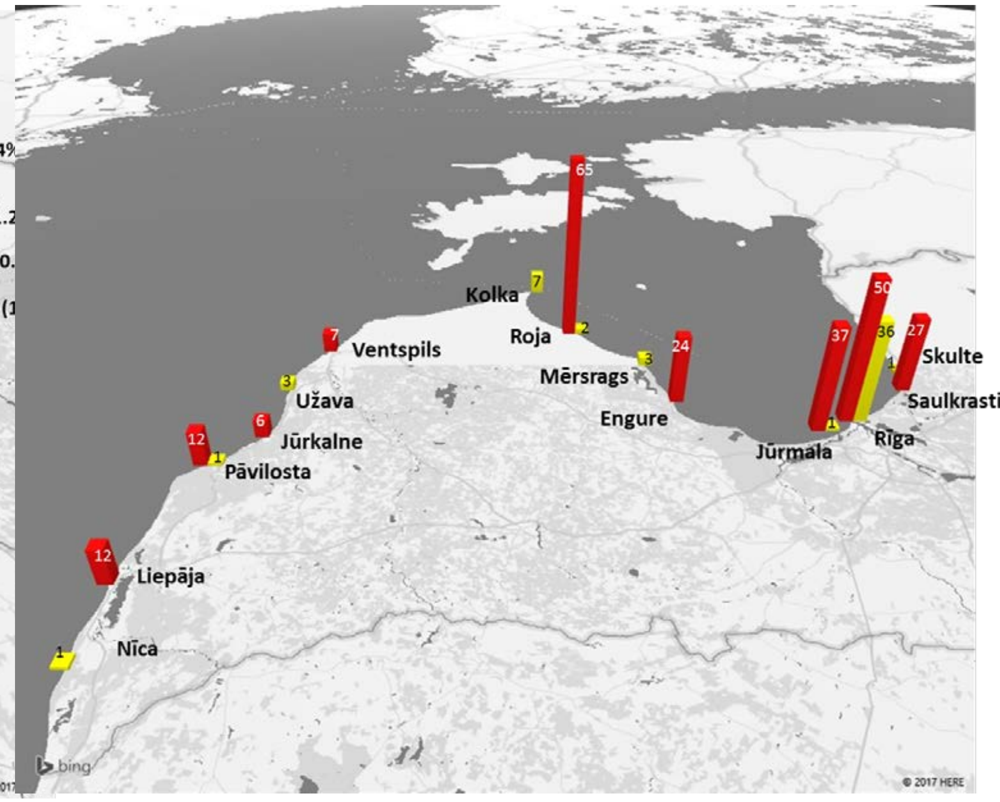

Figure 4.2. Number of seals killed in fishing gear from questioned fishermen (yellow bars) and number of seals washed ashore (red bars) from information collected by Nature Conservation Agency in 2016.

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5 ToR D. Update the database for seals

5.1 Historical context

In 2008, the WGMME recommended that a database be created for harbour and grey seal population indices within the ICES area to help ICES meet the requirements of its member countries and international organisations (e.g. OSPAR, NAMMCO, HELCOM). The recommendation was not a result of a formal request for advice from any of the above organisations but an attempt to collate salient information to facilitate the future work of the Working Group (WG). The initial proposal focused on Norway, Sweden, Denmark, Finland, Estonia, Russia, Germany, the Netherlands, Belgium, France, UK, and Ireland with an aim to eventually extend the area to include the Faroe Islands, the Barents Sea (Russia) and the Northwest Atlantic (Iceland, Greenland, Canada and the USA) (ICES, 2008).

In 2009, a simple MS Excel workbook was created and a more formal structure proposed. The WG reviewed the current and known data available from harbour and grey surveys. Denmark, Germany, the Netherlands, Belgium and the UK contributed data on harbour seals adult moult counts, adult breeding season counts and pup production estimates. For grey seals it contained pup production estimates and adult moult or summer counts. It was noted that the longevity and usefulness of the database would be entirely dependent on the frequency with which it was updated, and the ability to keep unpublished and valuable data secure. Working Group members also noted the variation in survey methods and reporting between some countries (ICES, 2009).

By 2010 Ireland, Sweden and Norway had also contributed some data (ICES, 2010). There were no updates in 2011 or 2012 although the increasing relevance of such information within the context of developing international marine mammal assessments (e.g. Ecosystem Quality Objectives) was stressed. In 2013 the database was updated and extended to include data from the Faroes, the Baltic, Russia, Iceland, Canada and the USA for grey seals and from Ireland, France, Norway, Sweden, Denmark, Iceland, Greenland and Canada for harbour seals. The updated database was submitted to the ICES data manager for secure future storage. The WG highlighted the need for the collation of seal population information to fulfil data requirements for population assessments under the Marine Strategies Framework Directive (MSFD) but noted that the database did not yet contain sufficient data to allow completion of an assessment of the proposed seal indicators (ICES, 2013).

In 2014, the WGMME reported on development of separate marine mammal databases being compiled by the USA (through the Atlantic Marine Assessment programme) and by Baltic countries through HELCOM. The original intention for the database to act as a repository for seal data across the entire ICES area was felt to be too ambitious and the WG proposed that the seal database should be revised such that it meets the MSFD reporting requirements for the OSPAR region only. Once again, the need to update the database with recent survey data was reiterated. The WG reviewed the usefulness of the database for fulfilling the needs of Member States under the MSFD in 2015; the information contained in the database at that time was not sufficient to enable a quantitative assessment of seal population abundance and distribution (MSFD indicator M-3) in the OSPAR region. The WG recommended that the relevant authorities from the OSPAR area provide data to populate the seal database at a time requested by the OSPAR Commission (ICES, 2014).

Later in 2015 OSPAR issued a formal data call to its Contracting Parties to submit data to support the assessment of MSFD common indictors for seals: M-5 grey seal pup production and M-3 seal population abundance and distribution. The format for this submission was different from the original MS Excel workbook proposed by WGMME (2008) because more information was necessary to support the MSFD requirements. Data submitted by Contracting Parties as part of the OSPAR formal data-call process formed the basis of draft assessments of indicators M-5 and M-3 for OSPAR's Intermediate Assessment (due in 2017).

In 2016, the OSPAR marine mammal expert group expressed a need for a central regional database to feed regional assessments of OSPAR common indicators on seals and the Biodiversity Committee (BDC) outlined a formal specification for such a database for both seals and seabirds to be built and hosted by ICES (OSPAR, 2016). Contracting Parties were requested to submit data through a web-based application to be developed as part of the database in response to an annual data call initiated by the OSPAR Secretariat. It is envisioned that the data submitted in response to the first OSPAR seal data call in 2015 will form the basis of this 'ICES-hosted OSPAR biodiversity Database for seabird and seal data' although these data have not yet been submitted to the ICES Data Centre.

The efforts in 2015 and 2016 of OSPAR to formally request data from Contracting Parties for the purpose of fulfilling MSFD requirements have led to some confusion as to the identity of 'the seal database' mentioned in past WGMME reports. The following sections seek to clarify the overlap and differences between the two data sources, and to highlight issues pertaining to their continued development.

5.2 'ICES seal database'

5.2.1 Area of relevance

The original intent of the WGMME proposal in 2008 was to create a central repository for data on the harbour (common) seal, *Phoca vitulina*, and the Atlantic grey seal, *Halichoerus grypus*, in particular numbers reported under national monitoring programmes. The idea was to collate information across ICES areas so that it was easier to access regional data incorporating seal numbers from several countries' coastlines. The scientific justification for this was that, as mobile marine predators, grey and harbour seals transit across national borders. Ecologically, there is merit in the WGMME knowing about trends in abundance of the two species where they co-occur and in documenting expansions and/or contractions in specific areas, especially at the outer extent of their range. The area of relevance is focused on the Northeast Atlantic and the North Sea (relevant countries include Denmark, Germany, the Netherlands, Belgium, UK, Norway, Sweden, Belgium, France and Ireland). Discussions also covered extension of the database to the Faroe Islands, the Baltic Sea in conjunction with the HELCOM Expert Group on Seals (i.e. to include the Baltic countries: Sweden, Finland, Russia, Estonia, Latvia, Lithuania and Poland and Russia), the Barents Sea (Russia) and the Northwest Atlantic (Iceland, Greenland, Canada and the USA) although few datapoints from these countries have been included to date.

5.2.2 Current status

The database is a MS Excel workbook with worksheets for:

- Harbour seal metadata;
- Harbour seal moult;

- Harbour seal breeding;
- Grey seal metadata;
- Grey seal pup numbers.

Numbers are available between 1986 and 2014 for some countries (primarily the UK). This year the WG provided updated numbers where available under ToR (a). However, for most countries and years there are no data, either because the database was not updated or because annual surveys were not performed and there were no data available; considerable effort would be necessary to update this database with relevant information from each country listed, although a significant amount of processed data is publicly available from many areas (e.g. UK, Wadden Sea). It is the intention of the present WG to make the effort to update this database.

5.3 'OSPAR seal database'

5.3.1 Area of relevance

The OSPAR seal database now refers to the collection of data generated in 2015–2016 expressly for the purpose of fulfilling MSFD assessment criteria; this database is formally referred to as the 'Biodiversity Data Portal: Seabird and seal abundance and distribution'. The area of relevance includes OSPAR Contracting Parties that are members of the European Union, and other European Economic Area countries participating in the MSFD assessment (e.g. Norway).

The assessments were performed at the scale of Assessment Units defined separately for harbour and grey seals and are summarized at the appropriate level of detail to allow assessment of abundance and distribution. The distributional aspect of the MSFD assessment is problematic (see ICES, 2016), and required that countries define subareas or haulout sites within their Assessment Units, within which the presence or absence of seals could be recorded. The geographical scale of this database is thus at a fairly high resolution. The Assessment Units in this database extend to coastlines of the UK, France, Belgium, Germany, the Netherlands, Denmark, and Norway, south of 62°N.

5.3.2 Current status

The OSPAR seal database is awaiting finalisation of an online web hosting and input mechanism to be delivered by the ICES Data Centre in 2017. Future annual OSPAR data calls will urge MSFD-participating countries to self-report via this system. Access will be restricted until concerns of data providers can be addressed in order to move towards the open access policy of OSPAR and the MSFD. It can be accessed at: http://ices.dk/marine-data/data-portals/Pages/Biodiversity.aspx

5.4 Future database concerns

The WG discussed whether it is necessary to maintain two seal databases and if the more recently collated OSPAR database would suffice. Members thought that the merit of attempting to maintain a less detailed, but geographically broader (e.g. including Iceland, Canada, USA) database for harbour and grey seals has scientific merit. If the database is updated at the annual WGMME meeting (e.g. either from publicly available sources online, or by direct contact with the data holders), a summary of seal population trajectories is thus easily accessible to WG members for the purposes of including up-to-date information in the annual report would be advantageous. This WGMME database (previously 'ICES seals database') should be held and maintained by the WG, under strict access control, for example by a single individual member.

The more detailed OSPAR database covering European waters will continue to be updated only via the formal OSPAR data call procedure, by responsible individuals in each Contracting Party.

5.5 Recommendation

The WGMME seeks clarity from the ICES and OSPAR Secretariats on the mechanism by which data on the abundance and distribution of seals across the ICES area (not just EU Member States) should be collated and stored, taking into consideration the issues raised above.

5.6 References

ICES. 2008. Report of the Working Group on Marine Mammal Ecology (WGMME). ICES, St Andrews, UK. ICES CM 2008/ACOM:44. 86 pp.

ICES. 2009. Report of the Working Group on Marine Mammal Ecology (WGMME). ICES, Vigo, Spain. ICES CM 2009/ACOM:21. 129 pp.

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ICES. 2014. Report of the Working Group on Marine Mammal Ecology (WGMME). ICES, Woods Hole, Massachusetts, USA. ICES CM 2014/ACOM:27. 234 pp.

OSPAR. 2016. Convention for the Protection of the Marine Environment of the North-East Atlantic Meeting of the Biodiversity Committee (BCD) (2016). Specification for the ICES hosted OSPAR biodiversity Database for seabird and seal data. Gothenburg. 29 February–5 March 2016.

6 ToR E. Update assessments of offshore cetaceans based on new results from the SCANS-III survey

At its 2016 meeting, WGMME collated available information on the distribution and abundance of offshore cetaceans under ToR e:

> To support OSPAR in the delivery of common indicator assessment of cetaceans through … an overview of data on cetacean species other than coastal bottlenose dolphins that may be available to make a regional assessment in the frame of indicator M-4.

WGMME was unable to fulfil the additional part of ToR e:

> To collate and assess the data identified … against the targets proposed.

The overview of the available information was compiled in Annex 6 of WGMME (2016). The key message from the overview was:

> Cetaceans are widely distributed, occur in a range of habitats and are, overall, abundant in OSPAR Regions II, III and IV. For most species there is only one robust estimate of abundance. For those species for which there are multiple estimates of abundance, the time-series are short relative to the life cycle of the species and the precision of the estimates is generally low leading to poor power to detect trends from these data. It is therefore not possible to infer with any confidence whether populations are decreasing, stable or increasing. However, there has been a clear shift in harbour porpoise distribution from north to south in the North Sea. Notwithstanding the inability to detect trends [based on currently available data], recent estimates of abundance are either similar to or larger than comparable earlier estimates. Despite the multiple pressures and threats facing cetaceans in this region, with the data available, there is currently no evidence of an impact of anthropogenic activity on either distribution or abundance of cetacean species in OSPAR Regions II, III and IV. More data are needed to make an informed assessment; results from a large-scale survey in summer 2016 will aid this process.

This year (2017), WGMME had available new information on the distribution and abundance of offshore cetaceans from the SCANS-III survey conducted in summer 2016 (Hammond *et al*., 2017).

The SCANS-III survey area and searching effort achieved is shown in Figure 6.1. The survey area was similar to the combination of SCANS-II in 2005 and CODA in 2007 (Hammond *et al*., 2013; CODA, 2009), but excluded waters to the south, west and north of Ireland (which were surveyed in summer 2016 as part of Irish project ObSERVE) and included for the first time coastal Norwegian waters from 62°N north to Vestfjorden. As in CODA in 2007, offshore Portuguese waters were not surveyed (see Figure 6.2). In addition to the above-mentioned surveys, the SAMM surveys covered the Channel and French waters of the Bay of Biscay in summer 2012 (Laran *et al*., 2017).

Shelf waters were surveyed entirely by air (except for the Kattegat and Belt Seas) using seven aircraft, in contrast to the mainly ship coverage in SCANS-II in 2005. The flexibility of aerial survey allowed good coverage of these waters with the exception of block J, where military restrictions and poor weather resulted in poor coverage. Three ships surveyed the Skagerrak, Kattegat and Belt Seas, and off-shelf waters west of Scotland, in the Bay of Biscay and west of Spain. Ship block 1 (Skagerrak) was also surveyed by air because of poor coverage by ship resulting from bad weather.

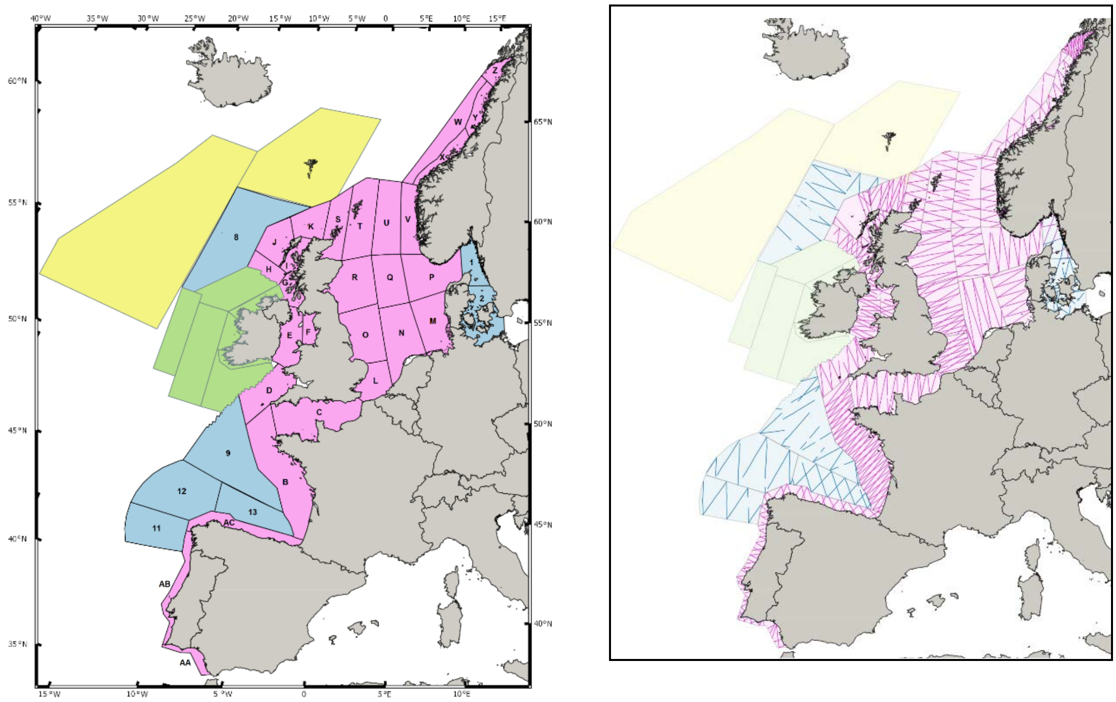

Figure 6.1. Survey blocks and total search effort achieved under all conditions in the SCANS-III aerial (pink) and ship (blue) survey. Not shown are blocks SVG and TRD, which covered parts of Norwegian fjords Bognafjord (near Stavanger) and Trondheim Fjord, respectively. Ship block 1 was also surveyed by air. Yellow blocks around the Faroe Islands and to the west were surveyed by the Faroe Islands as part of the North Atlantic Sightings Survey (NASS) in summer 2015. Green blocks to the south, west and north of Ireland comprise the Irish project ObSERVE aerial survey area.

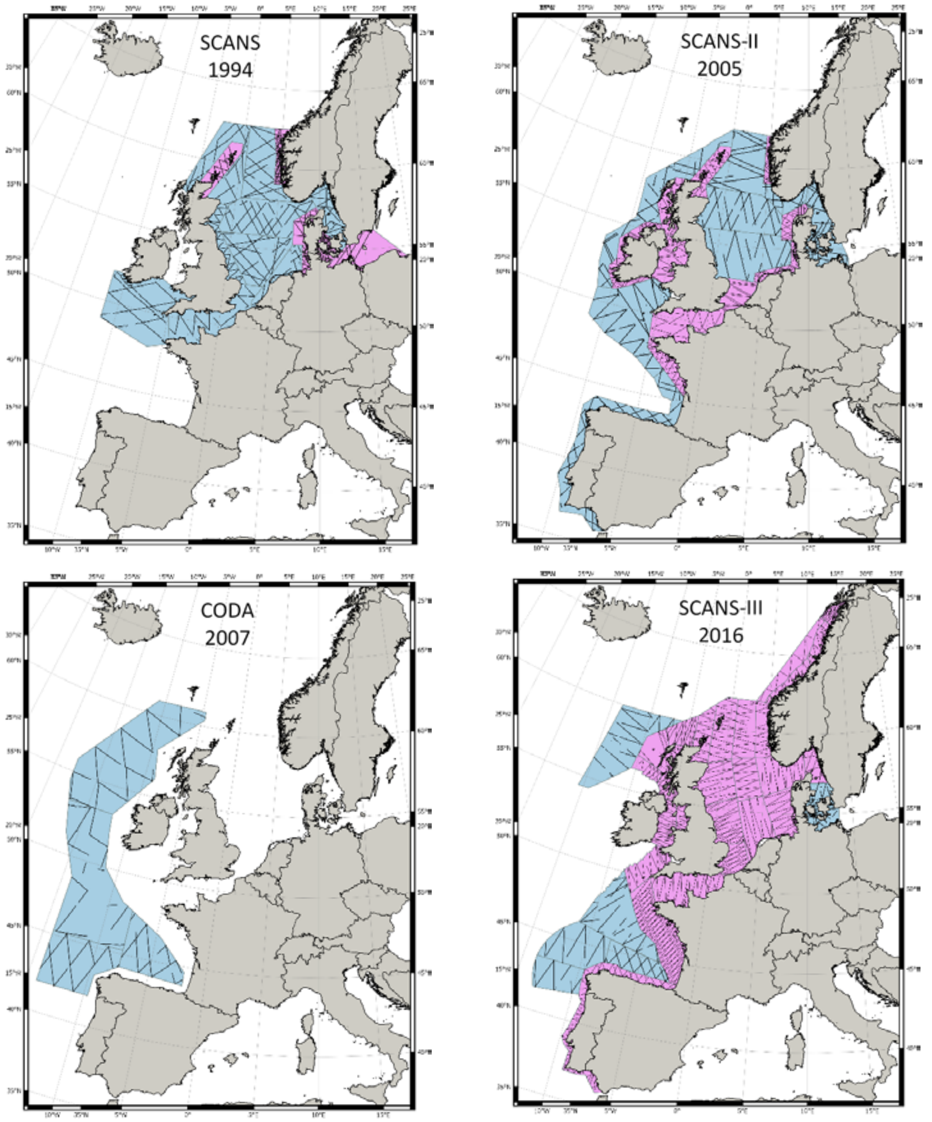

Figure 6.2. Areas surveyed, with on effort transect lines, by SCANS in 1994 (top left), SCANS-II in 2005 (top right), CODA in 2007 (bottom left) and SCANS-III in 2016 (bottom right).

Data collection and analysis methods used in the SCANS-III aerial and ship surveys in 2016 were the same as used in SCANS-II (and CODA), as described in Hammond *et al*. (2013). Details are given in Hammond *et al.* (2017). In SCANS-III, the aerial circle-back method that allows estimation of the proportion of animals detected on the transect line, g(0), was used for the first time for dolphins and minke whale, as well as for harbour porpoise.

Information was available for twelve species/species groups: harbour porpoise *Phocoena phocoena*, common bottlenose dolphin *Tursiops truncatus*, Risso's dolphin *Grampus griseus*, white-beaked dolphin *Lagenorhynchus albirostris*, Atlantic white-sided dolphin *Lagenorhynchus acutus*, short-beaked common dolphin *Delphinus delphis*, striped dolphin *Stenella coeruleoalba*, common minke whale *Balaenoptera acutorostrata*, fin whale *Balaenoptera physalus*, long-finned pilot whale *Globicephala melas*, sperm whale, *Physeter macrocephalus* and beaked/bottlenose whales as a combined species group (Ziphiidae).

Figure 6.3 shows the distribution of sightings of these species encountered on the survey.

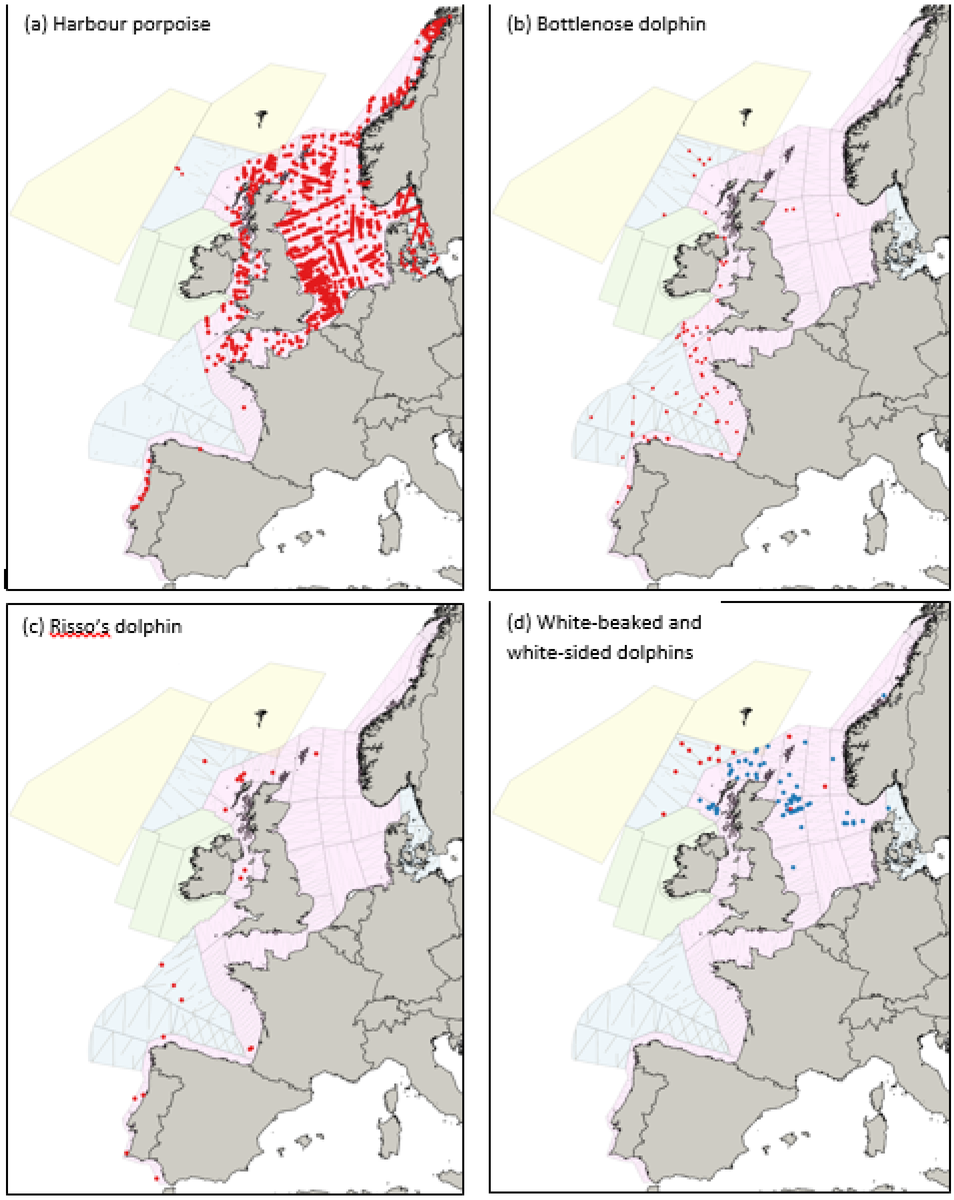

Figure 6.3. Distribution of sightings used in analysis of the most commonly detected species. Underlying effort is also that used in analysis: aerial survey - good and moderate conditions; ship survey - Beaufort 0–2 for harbour porpoise, Beaufort 0–4 for all other species. (a) harbour porpoise; (b) bottlenose dolphin; (c) Risso's dolphin; (d) white-beaked (blue dot) and white-sided (red dot) dolphins. Continued on following pages.

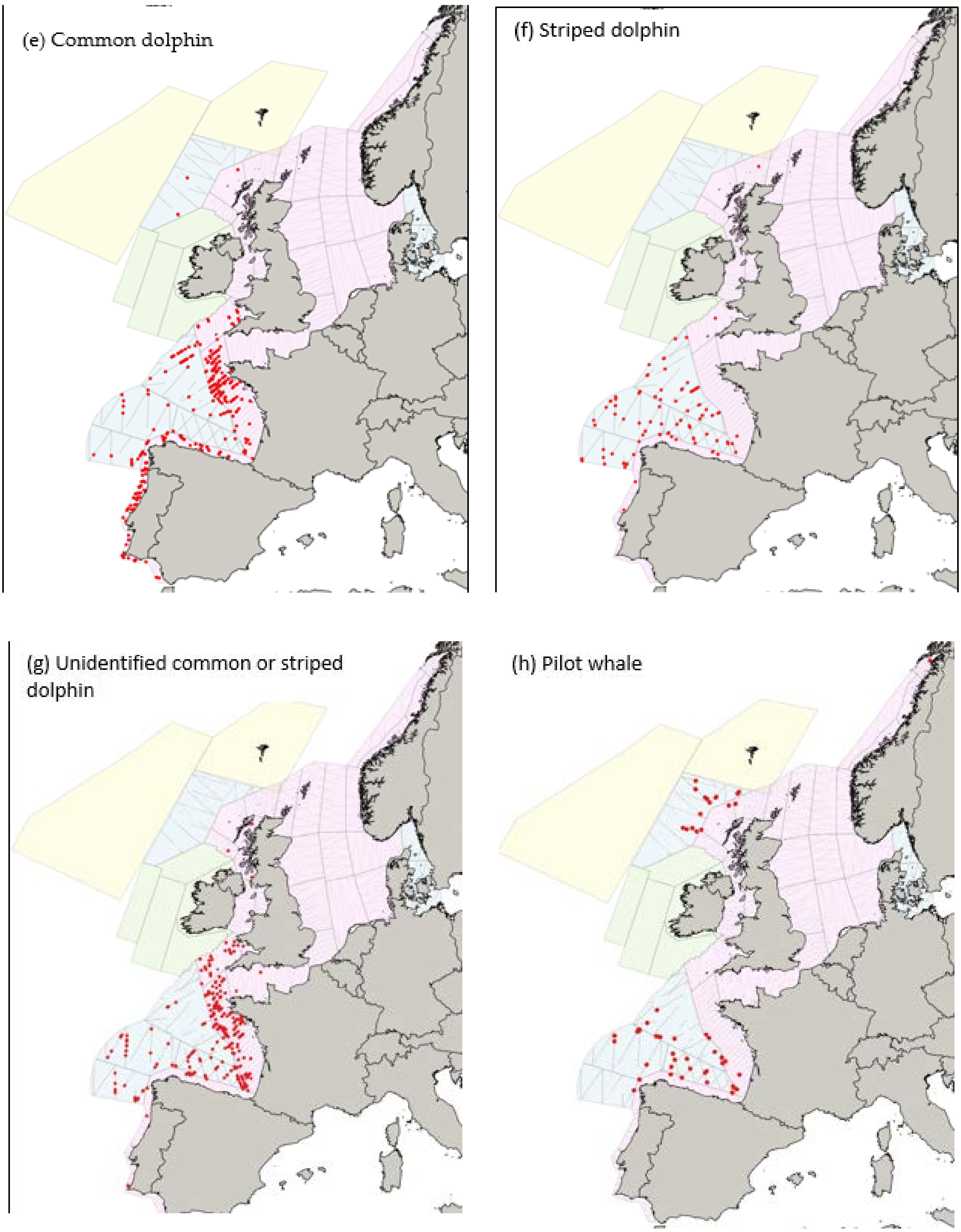

Figure 6.3. Continued. (e) common dolphin; (f) striped dolphin; (g) unidentified common or striped dolphin; (h) pilot whale.

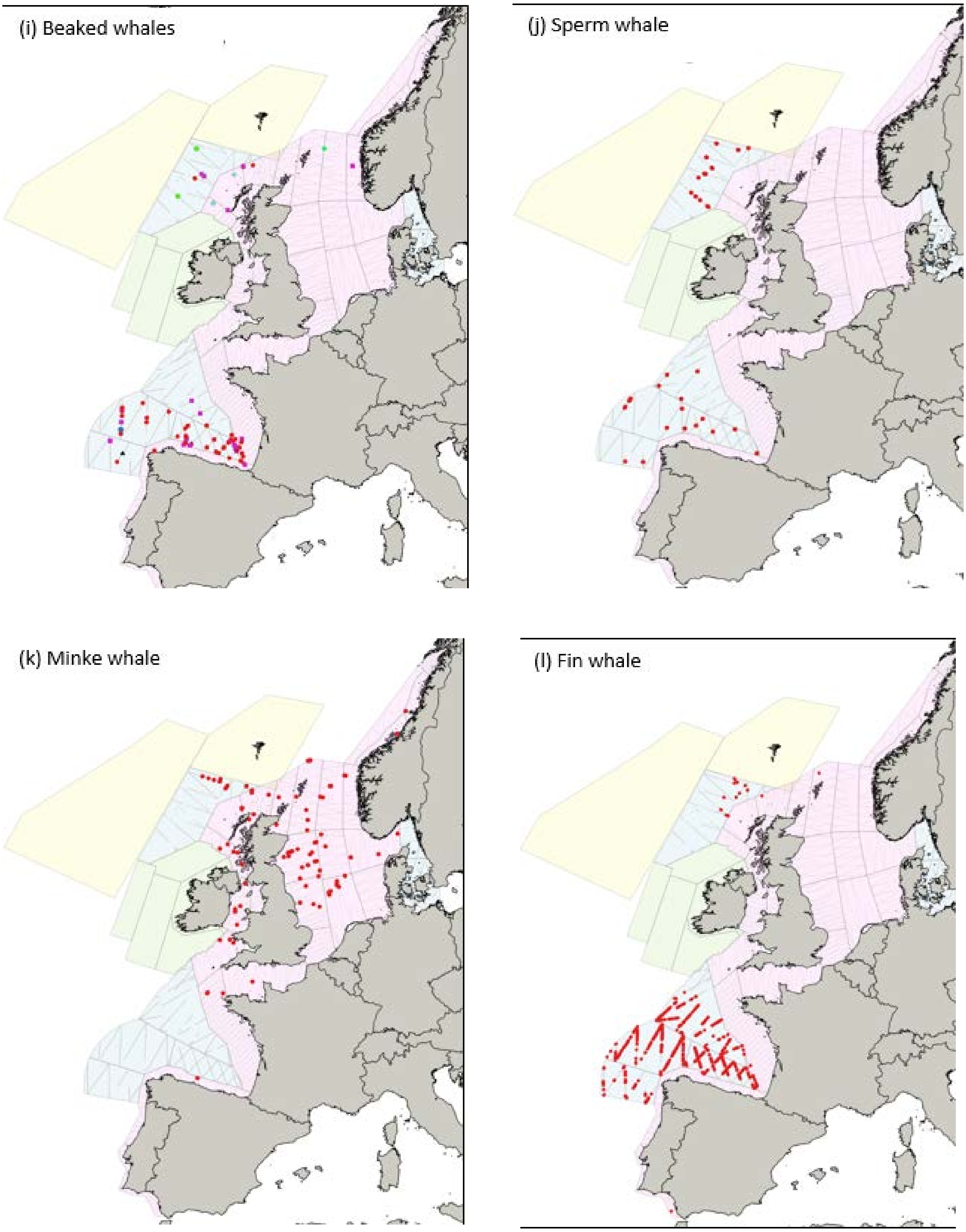

Figure 6.3. Continued. (i) beaked whales (Cuvier's beaked whale - red dot; Gervais beaked whale - blue dot; unidentified beaked whale - pink square; Unidentified Mesoplodon - black triangle; Sowerby's beaked whale - green dot; Northern bottlenose whale - turquoise dot); (j) sperm whale; (k) minke whale; (l) fin whale.

For those species with sufficient data, a low-resolution view of how abundance was distributed over the survey area can be seen from maps of estimated density by survey block. Maps for harbour porpoise; bottlenose, common and striped dolphin; and minke and fin whale are shown in Figure 6.4. Future work will include modelling of the new data from 2016 to investigate fine scale distribution and habitat use.

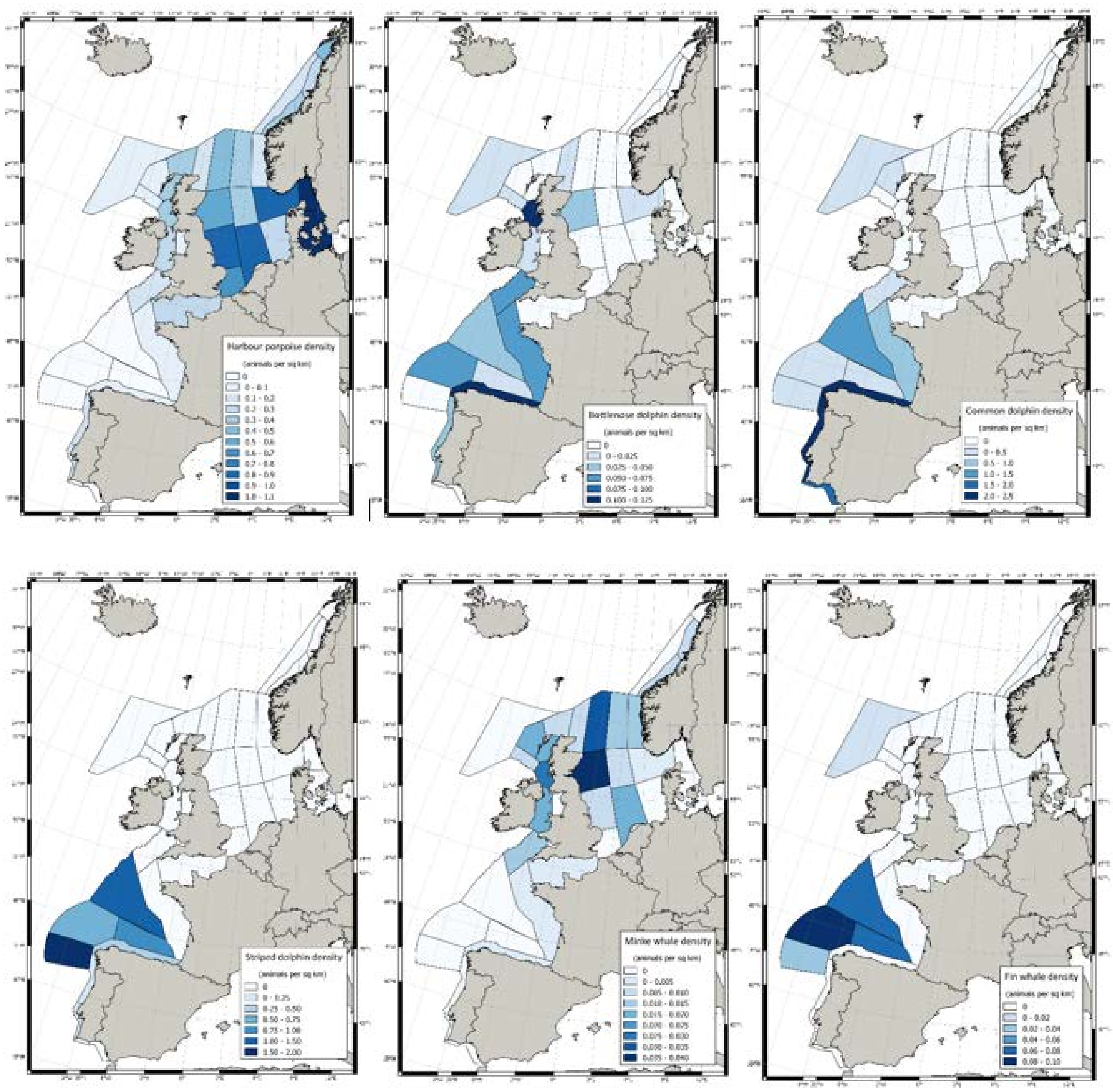

Figure 6.4. Estimated density from the SCANS-III survey in each survey block for harbour porpoise (top left), bottlenose dolphin (top middle), common dolphin (top right), striped dolphin (bottom left), minke whale (bottom middle) and fin whale (bottom right).

Table 6.1 gives the estimates of total abundance of each species for the whole survey area. Table 6.2 gives estimates of abundance for harbour porpoise in the five ICES assessment units and in Norwegian coastal waters.

Table 6.1. Estimates of total abundance in the whole SCANS-III survey area. CV is the coefficient of variation of abundance. CL low and CL high are the estimated lower and upper 95% confidence limits of abundance.

Species	Abundance	CV	CL low	CL high
Harbour porpoise	466 569	0.154	345 306	630 417
Bottlenose dolphin	27 697	0.233	17 662	43 432
Risso's dolphin	13 584	0.441	5943	31 047
White-beaked dolphin	36 287	0.290	18 694	61 869
White-sided dolphin	15 510	0.717	4389	54 807
Common dolphin	467 673	0.264	281 129	777 998
Striped dolphin	372 340	0.329	198 583	698 134
Unid common or striped	158 167	0.188	109 689	228 069
Pilot whale	25 777	0.345	13 350	49 772
Beaked whales (all species)	11 394	0.503	4494	28 888
Sperm whale	13 518	0.405	6181	29 563
Minke whale	14 759	0.327	7908	27 544
Fin whale	18 142	0.322	9796	33 599

Table 6.2. Estimates of harbour porpoise abundance in ICES Assessment Units, and Norwegian coastal waters north of 62°N. CV is the coefficient of variation of abundance. CL low and CL high are the estimated lower and upper 95% confidence limits of abundance. All estimates are from aerial survey except for the Kattegat and Belt Seas AU, which is from ship survey block 2. The sum of the estimates for the Celtic/Irish Seas and North Sea AUs (372 073) is slightly smaller than the sum of the contributing bocks (372 452) because block C, which spanned both AUs, was post-stratified in analysis.

Assessment Unit	Blocks	Abundance	CV	CL low	CL high
Celtic/Irish Seas	B, D–F, west C	26 700	0.25	16 055	42 128
North Sea	L–V, east C, SVG	345 373	0.18	246 526	495 752
West Scotland	G–K	24 370	0.23	15 074	37 858
Iberian peninsula	AA–AC	2898	0.32	1386	5122
Kattegat and Belt Seas	2	42 324	0.30	23 368	76 658
Norwegian coastal waters	W–Z, TRD	24 526	0.28	14 035	40 829

6.1 Consideration of new information

To ensure consistency across estimates of abundance from all SCANS surveys, it was necessary to reanalyse the ship survey data from SCANS and SCANS-II; details are given in Hammond *et al*. (2017). In addition, the new estimates of g(0) for dolphins and minke whale from SCANS-III aerial survey have been used to obtain revised aerial survey estimates for bottlenose, common and white-beaked dolphins and minke whale in 2005. These revised estimates are corrected for availability and perception bias. Previously published estimates were corrected only for availability, based on dive data from studies in other areas (Hammond *et al*., 2013). Table 6.3 includes these updated estimates.

6.1.1 Harbour porpoise

The observed distribution of harbour porpoises in 2016 (Figure 6.5) is similar to that observed in SCANS-II in 2005 (Hammond *et al*., 2013) but more sightings were made throughout the Channel in 2016 than previously. In 1994, no sightings were made in the Channel or the southern North Sea (Hammond *et al*., 2002). In 2005, there were a number of sightings at the far western end of the Channel (Hammond *et al*., 2013) and in 2012 there were sightings in both the western and eastern parts, but not the central part (SAMM (Suivi Aérien de la Mégafaune Marine/Aerial Monitoring of Marine Megafauna) surveys Laran *et al*., 2016). The progressive spread of sightings into most of the Channel over the past two decades indicates that harbour porpoise distribution has expanded, probably from the North Sea and the Celtic Sea, and now encompasses the entire Channel, at least in summer.

Total estimated abundance in 2016 for the area surveyed in 2005, i.e. excluding Norwegian coastal waters north of 62°N, is 442 000 (CV ≈ 0.15) compared to 375 000 (CV = 0.20) in 2005 (Hammond *et al*., 2013).

In the ICES AUs, the estimates in 2016 and 2005 are very similar in the Iberian Peninsula AU (2900, CV = 0.32 and 2880, CV = 0.72, respectively) and in the West Scotland AU (24 400, CV = 0.23 and 26 300, CV = 0.37). The southern part of the West Scotland AU was covered by the Irish ObSERVE project and information for this area is not yet available for 2016. In the Kattegat and Belt Seas AU, the estimate for 2016 of 42 000 (CV = 0.23) is consistent in terms of area surveyed only with the estimate for 2012 of 40 000 (CV = 0.24) (Viquerat *et al*., 2014). In the North Sea the estimate in 2016 (345 000, CV = 0.18) was similar to the estimate in 2005 (355 000, CV = 0.22; revised from Hammond *et al*., 2013) and 1994 (289 000, CV = 0.14; revised from Hammond *et al*., 2002), and to the model-based estimate using data from 2005–2013 of 361 000 (0.20) (Gilles *et al*., 2016).

Trends have been estimated for harbour porpoise in the North Sea and in the Skagerrak, Kattegat and Belt Seas, where there are three estimates of abundance from a comparable area (see Figure 6.5). Results show no support for changes in abundance since 1994.

The SCANS-III survey covered only a part of the Celtic and Irish Seas AU; the remaining part of the AU was covered by the Irish ObSERVE project, for which no estimate is available yet. It is thus not possible to present an estimate for this Assessment Unit at this time.

Table 6.3. Summary of population abundance estimates from large-scale surveys covering all or part of OSPAR areas II, III and IV (updated from Annex 6, Table B of WGMME 2016).

Species / Population	Survey / area	Estimate type	Estimate (CV)	Year	ICES WGMME AU	Notes	Reference
Beaked whales (all)	SCANS-II, CODA	Design-based	12,869 (0.31)	2005/2007	No specified AU	OSPAR areas II, III, IV & V	Rogan *et al*. (2017)
Beaked whales (all)	SCANS-III	Design-based	11,394 (0.50)	2016	No specified AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
Bottlenose dolphin	SCANS-II, CODA	Design-based	35,936 (0.21)	2005/2007	Oceanic waters	Part of AU, OSPAR area II, III, IV & V	Hammond *et al*. (2013); CODA (2009)
Bottlenose dolphin	SAMM / Channel, Biscay	Design-based	19,106 (0.23)	2011 (Winter)	Channel / Oceanic Waters	Channel + part Bay of Biscay	Laran *et al*. (2016)
Bottlenose dolphin	SAMM / Channel, Biscay	Design-based	13,255 (0.35)	2012	Channel / Oceanic Waters	Channel + part Bay of Biscay	Laran *et al*. (2016)
Bottlenose dolphin	SCANS-III	Design-based	27,697 (0.23)	2016	Oceanic waters	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
Common dolphin	SCANS-II, CODA	Design-based	174,485 (0.27)	2005/2007	Single AU	OSPAR areas II, III, IV & V	Hammond *et al*. (2013); CODA (2009)
Common dolphin	SCANS-III	Design-based	467,673 (0.26)	2016	Single AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
Common + striped	SCANS-II, CODA	Design-based	306,045(0.29)	2005/2007	Single AU	OSPAR areas II, III, IV & V	Hammond *et al*. (2013); CODA (2009)
Common + striped	SAMM / Channel, Biscay	Design-based	299,896 (0.11)	2011 (Winter)	Single AU	Channel + part Bay of Biscay	Laran *et al*. (in prep)
Common + striped	SAMM / Channel, Biscay	Design-based	696,013 (0.10)	2012	Single AU	Channel + part Bay of Biscay	Laran *et al*. (in prep)
Common + striped	SCANS-III	Design-based	998,180 (0.18)	2016	Single AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)

Species / Population	Survey / area	Estimate type	Estimate (CV)	Year	ICES WGMME AU	Notes	Reference
Fin whale	CODA, SCANS-II	Design-based	19,354 (0.24)	2005/2007	No specified AU	OSPAR areas II, III, IV & V	Macleod *et al*. (2011)
Fin whale	CODA, SCANS-II	Design-based	29,512 (0.26) including % of unid large whales	2005/2007	No specified AU	OSPAR areas II, III, IV & V	Macleod *et al*. (2011)
Fin whale	SCANS-III	Design-based	18,142 (0.38)	2016	No specified AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
Harbour porpoise	SCANS	Design-based	57,217 (0.52)	1994	Celtic and Irish Seas	Part of AU (SCANS block A and B2)	Revised from Hammond *et al*. (2002)
Harbour porpoise	SCANS-II	Design-based	107,344 (0.30)	2005	Celtic and Irish Seas	Part of AU (SCANS-II blocks B2, O, P, Q2, R2, W1, Z)	Revised from Hammond *et al*. (2013)
Harbour porpoise	SCANS-III	Design-based	26,700 (0.25)	2016	Celtic and Irish Seas	Part of AU (SCANS-III blocks B, C2, D, E, F)	Hammond *et al*. (2017)
Harbour porpoise	SCANS-II	Design-based	2,884 (0.72)	2005	Iberian Peninsula	Part of AU (SCANS-II block W2)	Revised from Hammond *et al*. (2013)
Harbour porpoise	SCANS-III	Design-based	2,898 (0.32)	2016	Iberian Peninsula	Part of AU (SCANS-III blocks AA, AB, AC)	Hammond *et al*. (2017)
Harbour porpoise	DEPONS	Model-based	372,167 (0.18)	2005-2013 (Spring)	North Sea	Area of prediction slightly smaller than AU	Gilles *et al*. (2016)
Harbour porpoise	DEPONS	Model-based	361,146 (0.20)	2005-2013 (Summer)	North Sea	Area of prediction slightly smaller than AU	Gilles *et al*. (2016)

Species / Population	Survey / area	Estimate type	Estimate (CV)	Year	ICES WGMME AU	Notes	Reference
Harbour porpoise	DEPONS	Model-based	228,913 (0.19)	2005-2013 (Autumn)	North Sea	Area of prediction slightly smaller than AU	Gilles *et al*. (2016)
Harbour porpoise	SCANS	Design-based	289,150 (0.14)	1994	North Sea	All of AU (SCANS blocks B1, J2, L, M, C, D1, E, F, G, H, Y)	Revised from Hammond *et al*. (2002)
Harbour porpoise	SCANS-II	Design-based	355,408 (0.22)	2005	North Sea	All of AU (SCANS-II blocks B1, H, J2, L, M, Q1, T, Y, V, Y)	Revised from Hammond *et al*. (2013)
Harbour porpoise	SAMM / Channel, Biscay	Design-based	31,199 (0.21)	2011 (Winter)	North Sea / Biscay & Iberia	Channel + part Bay of Biscay	Laran *et al*. (2016)
Harbour porpoise	SAMM / Channel, Biscay	Design-based	46,345 (0.12)	2012	North Sea / Biscay & Iberia	Channel + part Bay of Biscay	Laran *et al*. (2016)
Harbour porpoise	SCANS-III	Design-based	345,373 (0.18)	2016	North Sea	All of AU (SCANS-III blocks S, T, U, V, R, Q, P, O, M, N, L, C1)	Hammond *et al*. (2017)
Harbour porpoise	SCANS	Design-based	9,151 (0.24)	2005	W Scotland / N Ireland	Part of AU (SCANS blocks J1, D2)	Revised from Hammond *et al*. (2002)
Harbour porpoise	SCANS-II	Design-based	26,328 (0.37)	2005	W Scotland / N Ireland	Part of AU (SCANS-II blocks J1, N, Q3, R1)	Revised from Hammond *et al*. (2013)
Harbour porpoise	SCANS-III	Design-based	24,370 (0.23)	2016	W Scotland / N Ireland	Part of AU (SCANS-III blocks G, H, I, J, K)	Hammond *et al*. (2017)
Harbour porpoise	SCANS / Blocks I,X	Design-based	51,660 (0.30)	1994	Kattegat / Belt Seas	SCANS blocks I, X (includes Skaggerak north of AU)	Revised from Hammond *et al*. (2002)

Species / Population	Survey / area	Estimate type	Estimate (CV)	Year	ICES WGMME AU	Notes	Reference
Harbour porpoise	SCANS-II / Block S	Design-based	27,901 (0.39)	2005	Kattegat / Belt Seas	SCANS-II block S (includes Skaggerak north of AU)	Revised from Hammond *et al.* (2013)
Harbour porpoise	Kattegat / Belt Seas	Design-based	40,475 (0.24)	2012	Kattegat / Belt Seas	Equivalent to AU	Viquerat *et al.* (2014)
Harbour porpoise	SCANS-III / block 2	Design-based	42,324 (0.30)	2016	Kattegat / Belt Seas	Most of AU (SCANS-III block 2)	Hammond *et al.* (2017)
Minke whale	SCANS	Design-based	9,685 (0.23)	1994	Single AU	OSPAR areas II & III	Revised from Hammond *et al.* (2002)
Minke whale	SCANS-II, CODA	Design-based	26,784 (0.35)	2005/2007	Single AU	OSPAR areas II, III, IV & V	Revised from Macleod *et al.* (2011)
Minke whale	SAMM / Channel, Biscay	Design-based	363 (1.02)	2011 (Winter)	Single AU	Channel + part Bay of Biscay	Laran *et al.* (2016)
Minke whale	SAMM / Channel, Biscay	Design-based	5,223 (0.33)	2012	Single AU	Channel + part Bay of Biscay	Laran *et al.* (2016)
Minke whale	SCANS-III	Design-based	14,759 (0.33)	2016	Single AU	OSPAR areas I, II, III, IV & V	Hammond *et al.* (2017)
Minke whale	Norwegian / North Sea only	Design-based	5,429 (0.34)	1989	Single AU	56-61°N	Schweder *et al.* (1997)
Minke whale	SCANS / North Sea only	Design-based	7,495 (0.15)	1994	Single AU	All of AU (SCANS blocks C, D1, E, F, G)	Revised from Hammond *et al.* (2002)
Minke whale	Norwegian / North Sea only	Design-based	20,294 (0.26)	1995	Single AU	56-61°N	Schweder *et al.* (1997)
Minke whale	Norwegian / North Sea only	Design-based	11,713 (0.29)	1998	Single AU	56-61°N	Skaug *et al.* (2004)
Minke whale	Norwegian / North Sea only	Design-based	6,246 (0.48)	2004	Single AU	56-62°N	Solvang *et al.* (2015)

Species / Population	Survey / area	Estimate type	Estimate (CV)	Year	ICES WGMME AU	Notes	Reference
Minke whale	SCANS-II / North Sea only	Design-based	9,890 (0.34)	2005	Single AU	All of AU (SCANS-II blocks B1, H, J2, L, M, Q1, T, U, V, Y)	Revised from Hammond *et al*. (2013)
Minke whale	Norwegian / North Sea only	Design-based	6,891 (0.31)	2009	Single AU	56-62°N + 53-56°N west of 4°E	Solvang *et al*. (2015)
Minke whale	SCANS-III / North Sea only	Design-based	8,854 (0.24)	2016	Single AU	All of AU (SCANS-III blocks C1, T, U, V, R, Q, P, O, N, M, L)	Hammond *et al*. (2017)
Pilot whale	SCANS-II, CODA	Design-based	123,732 (0.35)	2005/2007	No specified AU	OSPAR areas II, III, IV & V	Rogan *et al*. (2017)
Pilot whale	SCANS-III	Design-based	25,777 (0.35)	2016	No specified AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
Sperm whale	CODA	Design-based	2,569 (0.26)	2007	No specified AU	OSPAR areas II, III, IV & V	Rogan *et al*. (2017)
Sperm whale	CODA	Design-based	5,623 (0.32) including % of unid large whales	2007	No specified AU	OSPAR areas II, III, IV & V	Rogan *et al*. (2017)
Sperm whale	SCANS-III	Design-based	13,518 (0.41)	2016	No specified AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
Striped dolphin	CODA	Design-based	61,364 (0.93)	2007	Single AU	Part of AU, OSPAR areas III & IV	CODA (2009)
Striped dolphin	SCANS-III	Design-based	372,340 (0.33)	2016	Single AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
White-beaked dolphin	SCANS	Design-based	23,716 (0.30)	1994	Single AU	Excluding western part of AU	Revised from Hammond *et al*. (2002)

Species / Population	Survey / area	Estimate type	Estimate (CV)	Year	ICES WGMME AU	Notes	Reference
White-beaked dolphin	SCANS-II	Design-based	37,689 (0.36)	2005	Single AU	OSPAR areas II, III & IV	Revised from Hammond *et al*. (2013)
White-beaked dolphin	SCANS-III	Design-based	36,287 (0.29)	2016	Single AU	OSPAR areas I, II, III, IV & V	Hammond *et al*. (2017)
White-beaked dolphin	SCANS / North Sea only	Design-based	22,619 (0.23)	1994	Single AU	All of AU (SCANS bocks B1, J2, L, M, C, D1, E, F, G, H, Y)	Revised from Hammond *et al*. (2002)
White-beaked dolphin	SCANS-II / North Sea only	Design-based	29,010 (0.35)	2005	Single AU	All of AU (SCANS-II blocks B1, H, J2, L, M, Q1, T, U, V, Y)	Revised from Hammond *et al*. (2013)
White-beaked dolphin	SCANS-III / North Sea only	Design-based	20,453 (0.36)	2016	Single AU	All of AU (SCANS-III blocks C1, T, U, V, R, Q, P, O, N, M, L)	Hammond *et al*. (2017)

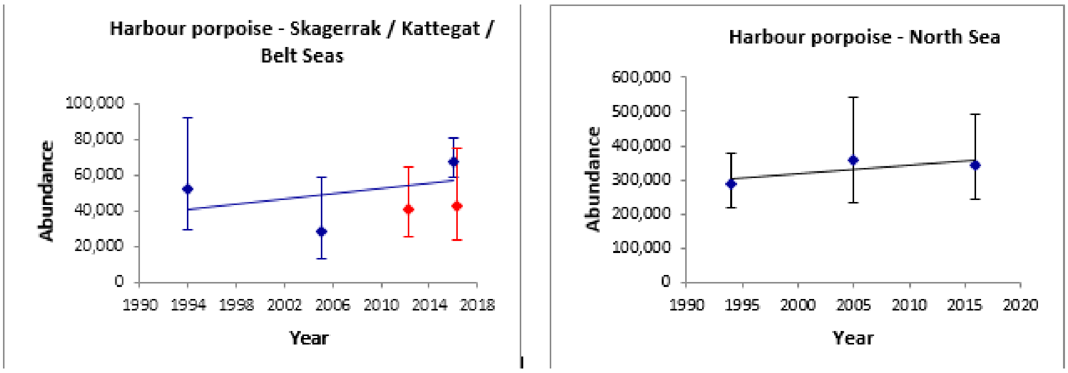

Figure 6.5. Trend lines fitted to time-series of three estimates of harbour porpoise abundance. Left: harbour porpoise in the Skagerrak/Kattegat/Belt Seas area (blue dots and line) – estimated rate of annual change = 1.24% (95%CI: -39; 67%), p = 0.81. Estimates for the Kattegat/Belt Seas population area are shown as red dots. Right: harbour porpoise in the North Sea – estimated rate of annual change = 0.8% (95%CI: -6.8; 9.0%), p = 0.18. Error bars are lognormal 95% confidence intervals.

6.1.2 Bottlenose dolphin

The observed distribution of bottlenose dolphins in 2016 was similar to that observed in SCANS-II and CODA in 2005/07 (Hammond *et al.*, 2013; CODA 2009) but most of the offshore sightings in 2007 were made in the ObSERVE survey area, for which information for 2016 is not yet available. The preliminary estimate of abundance for 2016 of 27 700 (CV = 0.23) is smaller than that from 2005/07 of 35 900 (CV = 0.21) (WGMME 2017) but a direct comparison between estimates for 2016 and 2005/07 should not be made until estimates are available for equivalent areas.

6.1.3 White-beaked dolphin

The observed distribution of white-beaked dolphins in 2016 is similar to that observed in SCANS-II in 2005 (Hammond *et al.*, 2013) and in SCANS in 1994 (Hammond *et al.*, 2002). The estimate of abundance in 2016 of 36 300 (CV = 0.29) is very similar to the estimate from SCANS-II in 2005 of 37 700 (CV = 0.36) (revised from Hammond *et al.*, 2013) but higher than the estimate from SCANS in 1994 of 22 600 (CV = 0.23) (revised from Hammond *et al.*, 2002).

A trend has been estimated for white-beaked dolphin in the North Sea, where there are three estimates of abundance from a comparable area (see Figure 6.6). Results show no support for changes in abundance since 1994.

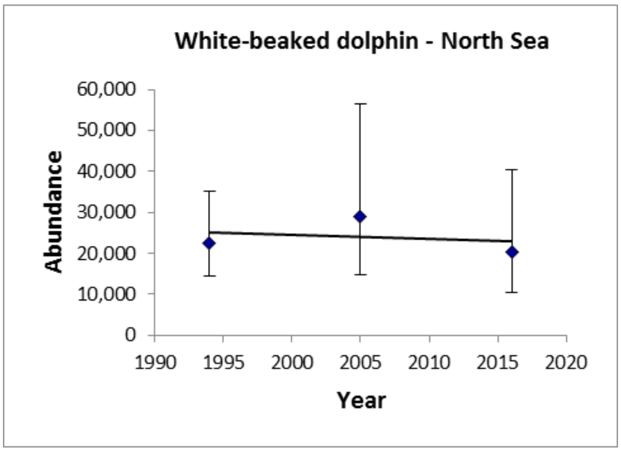

Figure 6.6. Trend line fitted to time-series of estimates of white-beaked dolphin abundance in the North Sea. Estimated rate of annual change = -0.5% (95%CI: -18; 22%), p = 0.36. Error bars are lognormal 95% confidence intervals.

6.1.4 Common and striped dolphins

The observed distributions of common and striped dolphins in 2016 are similar to those observed in SCANS-II and CODA in 2005/07 (Hammond *et al.*, 2013; CODA 2009) and in the SAMM surveys in the Channel and French waters of the Bay of Biscay in summer 2012 (Laran *et al.*, 2016). Some sightings in 2005 and 2007 were made in the ObSERVE survey area, in which information for 2016 is not yet available. The distribution of common dolphins appears to be strongly concentrated in shelf waters but a substantial number of unidentified common or striped dolphin sightings were also made in offshore waters, at least some of which were likely to have been common dolphins. Striped dolphins appear to be strongly concentrated in offshore waters but some of the unidentified sightings in shelf waters could have been striped dolphins.

The estimates of abundance in 2016 of 468 000 (CV = 0.26) common dolphin, 372 000 (CV = 0.33) striped dolphin and 158 000 (CV = 0.19) unidentified common or striped dolphins sum to almost one million animals. These estimates are substantially larger than the estimates for 2005/2007 of 174 000 (CV = 0.27) common dolphin and 61 400 (CV = 0.93) striped dolphin, respectively (revised from Hammond *et al.*, 2013; CODA 2009). A direct comparison between estimates for 2016 and 2005/07 should not be made until estimates are available for equivalent areas.

However, the estimate of common and striped dolphins in summer 2012 from the SAMM surveys in the Channel and French waters in the Bay of Biscay was around 700 000 animals (Laran *et al.*, 2016). The SAMM survey area did not include Spanish waters that were included in SCANS-III in 2016 and the estimate was not corrected for animals missed on the transect line. The estimates from SCANS-III in 2016 and SAMM in 2012 therefore appear to be compatible.

6.1.5 Long-finned pilot whale

The observed distribution of pilot whales was similar in 2016 to that observed in SCANS-II and CODA in 2005/07 (Rogan *et al.*, 2017) but the majority of the sightings in 2007 were made in the ObSERVE survey area, for which information for 2016 is not yet available. The absence of information from Irish waters may partly explain why the estimate of abundance for 2016 of 25 800 (CV = 0.35) is considerably smaller than that from 2005/07 of 124 000 (CV = 0.35) (Rogan *et al.*, 2017) but a direct comparison should not be made until estimates are available for equivalent areas.

6.1.6 Beaked whales (all species)

The observed distribution of beaked whales was similar in 2016 to that observed in CODA in 2007 (CODA 2009) and from opportunistic sightings (WGMME 2016). Some of these sightings were made in the ObSERVE survey area, for which information for 2016 is not yet available.

The estimate of abundance of all beaked whale species combined for 2016 of 11 400 (CV = 0.50) is similar to the equivalent estimate from SCANS-II and CODA in 2005/2007 of 12 900 (CV = 0.31) (Rogan *et al.*, 2017) but a direct comparison should not be made until estimates are available for equivalent areas.

6.1.7 Sperm whale

The observed distribution of sperm whales was similar in 2016 to that observed in CODA in 2007 (Rogan *et al.*, 2017). Some of these sightings were made in the ObSERVE survey area, for which information for 2016 is not yet available.

The estimate of abundance of sperm whales in 2016 of 13 500 (CV = 0.41) is larger than both the estimate from CODA in 2007 of 2600 (CV = 0.26) for identified sperm whales and the estimate of 5600 (CV = 0.32) if a proportion of unidentified large whales is included (Rogan *et al.*, 2017). However, a direct comparison should not be made until estimates are available for equivalent areas.

6.1.8 Minke whale

Between 1994 and 2005 there was some evidence that minke whale distribution in the North Sea had shifted to the south (Hammond *et al.*, 2013). The observed distribution of minke whale in 2016 was similar to that observed in 2005 in the North Sea, and similar overall to that in 2005/07 (Hammond *et al.*, 2013; Hammond *et al.*, 2011). However, many sightings in 2007 were made in the Irish ObSERVE survey area, for which information from 2016 is not yet available.

The estimate of abundance in 2016 of 14 800 (CV = 0.33) is smaller than the estimate for 2005/07 from SCANS-II and CODA of 26 800 (CV = 0.35) (revised from Hammond *et al.*, 2011). This may be partly because of the lack of an estimate in Irish waters but a direct comparison should not be made until estimates are available for equivalent areas. The estimate for 2016 in the North Sea was 8900 (CV = 0.24), which is within the range of previous estimates from SCANS, SCANS-II and Norwegian surveys.

A trend has been estimated for minke whale in the North Sea, where there are eight estimates of abundance from a comparable area (see Figure 6.7); three from SCANS surveys and five from the Norwegian Independent Line Transect Surveys (NILS) (Bøthun *et al.*, 2009; Schweder *et al.*, 1997; Skaug *et al.*, 2004; Solvang *et al.*, 2015). All these estimates relate to the North Sea bounded to the north by 62°N but the earlier Norwegian estimates covered a smaller area, between 56°N and 61°N. The most recent Norwegian minke whale estimate for 2009 includes waters south to 53°N.

6.1.9 Fin whale

The observed distribution of fin whales in 2016 was similar to that observed in CODA in 2007 (Hammond *et al.*, 2011). The estimate of abundance in 2016 of 18 100 (CV = 0.38) is very similar to the estimate from 2007 of 19 300 (CV = 0.24) for identified fin whales but smaller than the estimate for 2007 that included a proportion of unidentified large whales of 29 500 (CV = 0.21) (Hammond *et al*., 2011). Analyses to account for unidentified large whales have not yet been undertaken for the SCANS-III data. The 2007 estimate also included waters to the west of Ireland, which SCANS-III did not, and a direct comparison should not be made until estimates are available for equivalent areas.

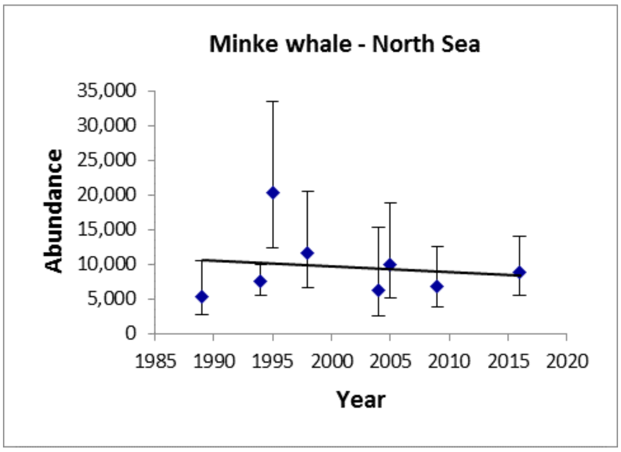

Figure 6.7. Trend line fitted to time-series of estimates of minke whale abundance in the North Sea. Estimated rate of annual change = -0.25% (95%CI: -4.8; 4.6%), p = 0.90. Error bars are lognormal 95% confidence intervals.

6.2 Power to detect trends

In any assessment of trend, it is important to consider the statistical power to detect a change in abundance of a given magnitude. Simple power analyses (ignoring additional variance from variation in the number of animals present in the area at the time of the survey) were conducted to determine the annual rate of decline that could be detected with high (80%) power from the available estimates of abundance. Power was calculated using the simplified inequality:

$$r^2n^3 > 12CV^2 (Z_{\alpha/2} + Z_\beta)^2$$

where r = rate of change over the time period in question, n = the number of surveys during the time period, CV = coefficient of variation of abundance, $Z_{\alpha/2}$ = the value of a standardised random normal variable for the probability of making a Type I error, α (set to 0.05), Z_β is the value of a standardised random normal variable for the probability of making a Type II error, β, and power is $(1-\beta)$ (Gerrodette, 1987).

Table D gives the results of the power calculations. The annual rates of decline that can be detected with 80% power from the three estimates in the North Sea are 1.8% for harbour porpoise and 5% for white-beaked dolphin. For minke whale, the eight estimates for the North Sea are quite variable but have 80% power to detect a 0.5% annual rate of decline.

Table 6.4. Results of power calculations to determine the annual rate of decline that could be detected by the available data with 80% power, n is the number of abundance estimates and CV = average CV of abundance for the available estimates.

Species	Region	n	CV	Annual rate of decline detectable at 80% power
Harbour porpoise	Skagerrak / Kattegat / Belt Seas	3	0.30	3.7%
Harbour porpoise	North Sea	3	0.18	1.8%
White-beaked dolphin	North Sea	3	0.36	5%
Minke whale	North Sea	8	0.30	0.5%

6.3 Concluding remarks - lessons learned from the SCANS experience

Overall, the results from these large-scale international surveys have greatly expanded our knowledge of the distribution and abundance of cetacean species in the European Atlantic, enabling bycatch and other anthropogenic stressors to be placed in a population context and giving a strong basis for assessments of conservation status. The information now available forms a good foundation for a large-scale time-series for the coming decades.

SCANS-type surveys as stand-alone projects require considerable resources focused at one point in time. However, considering their current decadal-scale frequency and the number of countries involved (around ten), the annual cost per country is small. Even if the frequency were increased to match EU Directive reporting cycles of six years, they should be readily affordable.

Although there have been three successful SCANS projects, they do not form a programme of surveys; each one has been developed from scratch by a team of dedicated scientists. If European Atlantic range states value the information provided by SCANS it would be more appropriate to future surveys to be driven by government agencies responsible for implementing national and European policy.

The results presented to date will be integral to cetacean assessments undertaken for OSPAR's Quality Status Report and for the Marine Strategy Framework Directive assessments of Good Environmental Status. The results also enable the impact of bycatch and other anthropogenic pressures on cetacean populations to be determined, fulfilling a suite of needs under the EU Habitats Directive and the Agreement on the Conservation of Small Cetaceans in the Baltic, Northeast Atlantic, Irish and North Seas (ASCOBANS). Estimates of absolute (unbiased) abundance are required for these tasks, at least periodically, and SCANS-type two-team survey methods are needed to achieve this (Hammond *et al*., 2017).

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7 ToR F. Contribute regional text to new ecosystem overviews for (i) Iceland, (ii) Norwegian Seas, (iii) Baltic, (iv) Azorean ecoregion and (v) the Oceanic Northeast Atlantic ecoregion

Figure 7.1 shows the ICES ecoregions and WGMME was tasked with contributing text on the status and trends of marine mammals inhabiting the Iceland, Norwegian Seas, Baltic, Azorean and the Oceanic Northeast Atlantic ecoregions.

As stated in the ICES Technical guidelines for ecosystem overviews (ICES, 2016), for each ecosystem component short descriptions should be provided about state and trends in addition to reporting on the anthropogenic pressures to which these species are subject in each ecoregion.

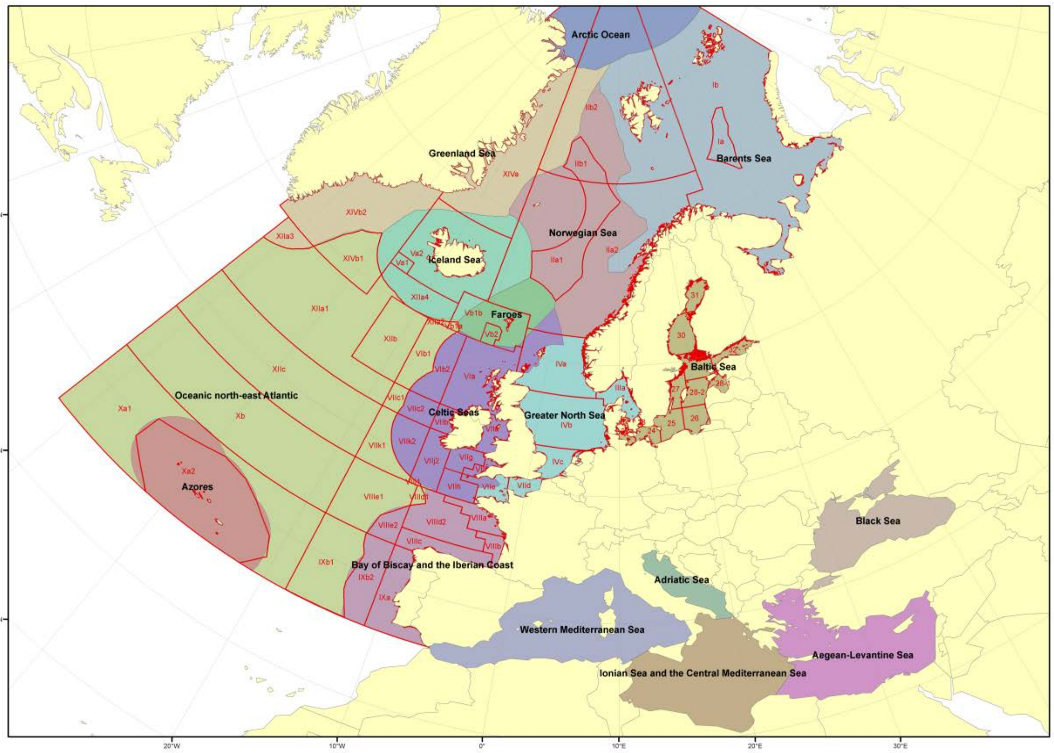

Figure 7.1. Map showing the ICES ecoregions (ICES, 2016).

7.1 ToR f (i) Iceland

Two pinniped species breed in Iceland: grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*). Four other species, namely harp seal (*Phoca groenlandica*), bearded seal (*Erignathus barbatus*), hooded seal (*Cystophora cristata*) and ringed seal (*Pusahispida*, formerly *Phocahispida*)), occur on a less regular basis. A total of 23 species of cetaceans has been seen Icelandic waters, twelve of which are seen on a regular basis: blue whales (*Balaenoptera musculus*), fin whales (*B. physalus*), sei whales (*B. borealis*), common minke whales (*B. acutorostrata*), humpback whales (*Megapterano vaeangliae*), sperm whales (*Physeter macrocephalus*), northern bottlenose whales (*Hyperoodona pullatus*), long-finned pilot whales (*Globicephala melaena*), killer whales (*Orcinus orca*), white-beaked dolphin (*Lagenorhynchus albirostris*), white-sided dolphins (*L. acutus*) and harbour porpoise (*Phocoena phocoena*).

Both the Icelandic harbour seal and grey seal populations are currently in decline. The harbour seal population has decreased from 33 000 animals in the first census in 1980 to the current (2016) estimate of 7700 animals. The largest observed decline, however, occurred between 1980 and 1989 when a bounty system was in effect, but the decline continues and the current estimated population size is the smallest that has ever been observed (Figure 7.2). The Icelandic grey seal population has been surveyed at irregular intervals since 1982, when the population size was estimated to be 9000 animals. The latest estimate from 2012 indicated a population size of 4200 animals. A new grey seal census is planned in 2017.

Large-scale cetacean surveys conducted at regular intervals between 1987 and 2016 have revealed varying trends in abundance (Pike and Lockyer, 2009). Humpback whales have shown high rates of increase and fin whale abundance also increased significantly during 1987–2001. Fin whale abundance in the Central North Atlantic increased significantly between 1987–2015 (Víkingsson *et al*., 2009), particularly in the Irminger Sea between Iceland and Greenland (Figure 7.3). Abundance of common minke whales has decreased substantially in Icelandic coastal waters since 2001, most likely due to decreased availability of important prey species such as sandeel (Ammodytidae) and capelin (*Mallotus villosus*) (Víkingsson *et al*., 2015).

The most important anthropogenic threat to marine mammals in Icelandic waters is bycatch of seals and small cetaceans.

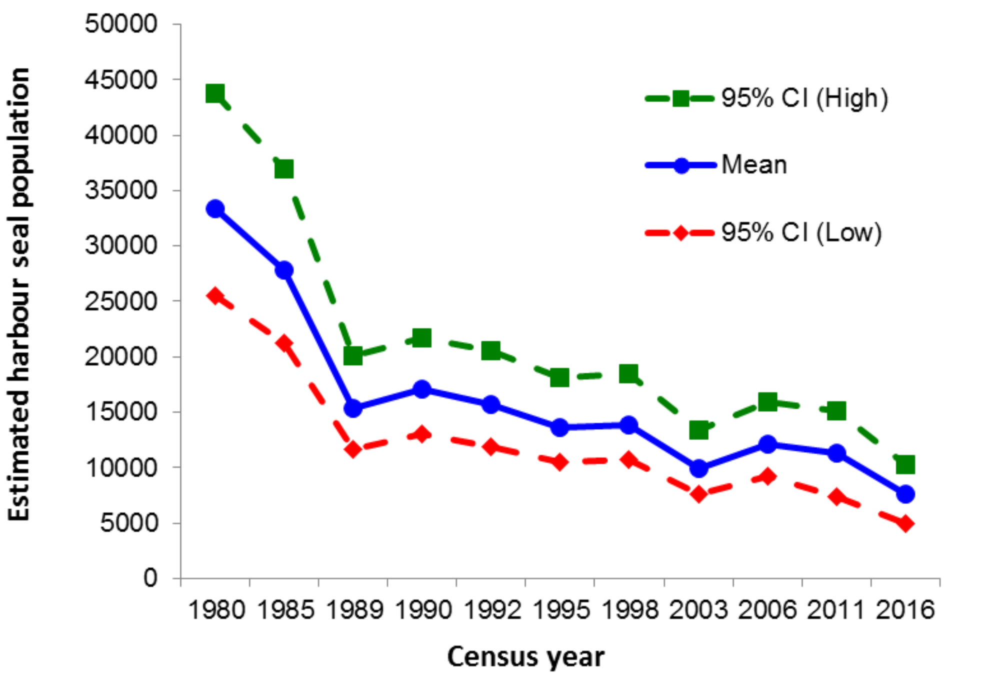

Figure 7.2. The trend in the Icelandic harbour seal population from 1980 to 2016. The mean values (blue) and 95% confidence intervals are shown.

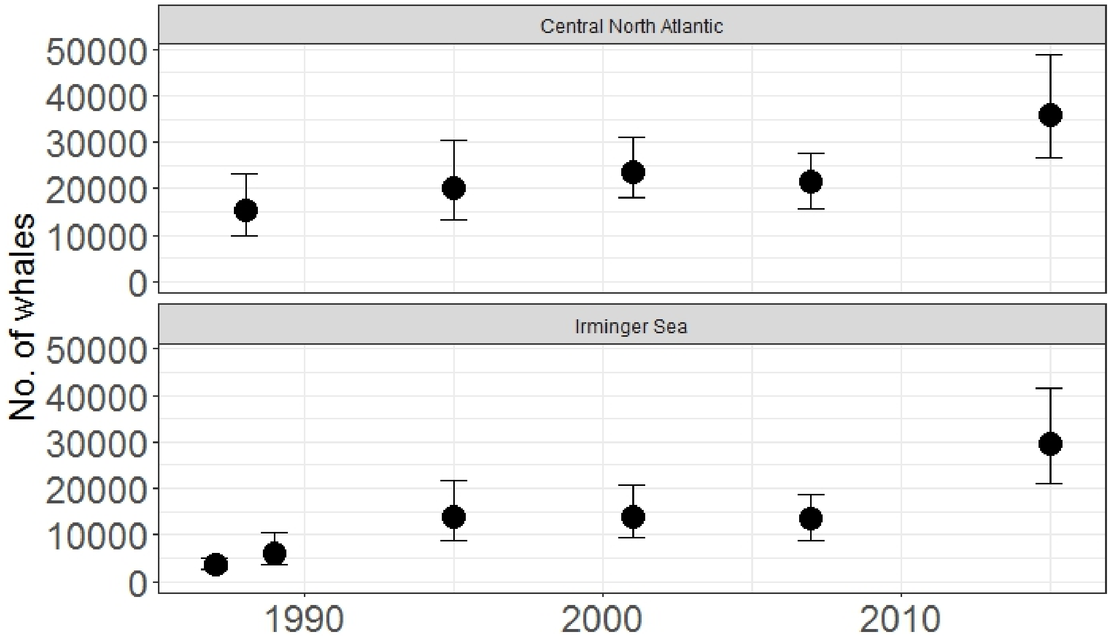

Figure 7.3. Abundance (with 95% confidence intervals) of fin whales according to the North Atlantic Sightings Surveys 1987–2001 in the total Central North Atlantic stock area (upper) and in the Irminger Sea west of Iceland (lower) (from Vikingsson *et al.*, 2015).

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7.2 ToR f (ii) Norwegian waters

Norwegian waters are home to a variety of seal and cetacean species. The Institute of Marine Research published a report on marine mammals in Norwegian waters in 2010 (Bjørge *et al.*, 2010), and the following description is largely based on this report.

Two species of seals are present year-round in coastal waters; harbours seals (*Phoca vitulina*) and grey seals (*Halichoerus grypus*). In addition, infrequent visitors to the coasts include bearded seals (*Erignathus barbatus*), harp seals (*Pagophilus groenlandicus*), hooded seals (*Cystophora cristata*), walrus (*Odobenus rosmarus*) and ringed seals (*Pusahispida*). In addition to the two coastal seal species, Norway is responsible for the monitoring and management of harp and hooded seals in the West Ice region between Jan Mayen and Eastern Greenland (see Figure 7.4), as well as for Arctic species in the waters around the Svalbard archipelago.

The abundance of harbour seals in central Norway has declined since the late 1990s, mainly because of hunting, but is at present recovering. Surveys of grey seals along the Norwegian coast have shown a reduction in pup production by between 50–60% between 2007–2008 and 2014–2015 in mid-Norway. The decline in the grey seal population is probably mainly due to increased bycatches in gillnet fisheries for monkfish and cod (IMR, 2016). A 2012 survey of harp and hooded seals in the Greenland Sea showed stable levels for these two species, with harp seals at a high level and hooded seals at a continued historical low level.

Figure 7.4. Satellite tracks of 18 hooded seals tagged in 2007–2008 in the pack ice region of the West Ice (East Greenland). Data were collected as part of the MEOP project (Marine Mammals Exploring the Oceans Pole to Pole) under the International Polar Year program. Data credit: Kit Kovacs & Christian Lydersen (Norwegian Polar Institute), Tore Haug (Institute of Marine Research) & Mike Fedak (Sea Mammal Research Unit).

Several cetacean species are commonly observed in Norwegian waters, either on a year-round basis or as seasonal visitors during the productive summer season. Year-round residents include minke whales (*Balaenopteraacuto rostrata*), white-beaked dolphins (*Lagenorhynchus albirostris*), white-sided dolphins (*L. acutus*) harbour porpoises (*Phocoena phocoena*), killer whales (*Orcinus orca*) and long-finned pilot whales (*Globicephala melaena*), while regular summer visitors include blue whales (*B. musculus*), fin whales (*B. physalus*), sei whales (*B. borealis*), humpback whales (*Megapteranovae angliae*), sperm whales (*Physeter macrocephalus*) and beluga whales (*Delphinapterus leucas*). Some of these latter species may be present year-round in smaller numbers, specifically non-breeding individuals who may remain within rich feeding grounds rather than undertake southward migrations to common breeding grounds.

Counts for Northeast Atlantic minke whales during 2007–2013 show a stable overall population level but there has been a general displacement of minke whales and other baleen whales towards the Northeast, implying a shift from the Norwegian Sea to the Barents Sea.

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7.3 ToR f (iii) Baltic

Three seal species, the grey seal (*Halichoerus grypus*), harbour seal (*Phoca vitulina*) and the ringed seal (*Pusa hispida*, Syn: *Phoca hispida*), and the harbour porpoise, occur on regular basis in the Baltic Sea (HELCOM, 2015). There are an additional thirteen cetacean species (e.g. humpback whale (*Megaptera novaeangliae*), minke whale (*Balaenoptera acutorostrata*), Sowerby's beaked whale (*Mesoplodon bidens*), killer whale (*Orcinus orca*), common bottlenose dolphin (*Tursiops truncatus*), white-beaked dolphin (*Lagenorhynchus albirostris*)) and one seal species that have been noted as rare or vagrants.

The distribution of seals differs depending on the species. The only species present in the whole Baltic is the grey seal (HELCOM, 2015). Its population growth rate is estimated to be around 7.9% (considering data from 2000–2014), and it is regarded by HELCOM as being in a "good environmental status" (HELCOM, 2016).

The ringed seal occurs mainly in the Bothnian Bay, Gulf of Finland, Archipelago Sea, Gulf of Riga and in Estonian coastal waters. It has been listed as "vulnerable" by the **International Union for Conservation of Nature (**IUCN) and the population growth rate is considered to be below "good environmental status" following the HELCOM assessment criteria. The population in the Gulf of Finland has been decreasing and currently a number of around 100 animals is estimated. The majority of animals (around 8000) are found in the Bothnian Bay (HELCOM, 2016).

The harbour seal mainly occurs in the southern Baltic and the population in this area (HELCOM, 2015) has an estimated growth rate of 8.4% (considering data from 2002–2014). The neighbouring Kalmarsund population provides influx (population growth rate in this area has been 9% since 1975) to the population in the southern Baltic (HELCOM, 2016). Harbour seals numbers have also been strongly affected by outbreaks of the Phocine Distemper Virus (PDV) in 1988 (mortality rate ~-50%) and 2002 (mortality rate ~-16%) (Härkönen *et al*., 2006).

The only regularly occurring and reproducing cetacean species in the Baltic is the harbour porpoise (*Phocoena phocoena*) (Koschinski, 2001). In the Baltic proper, a large population decline has been observed during the past 50–100 years (Skòra *et al*., 1988; Koschinski, 2001) and this population is listed as critically endangered under the IUCN red list and under HELCOM. 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).

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 is uncertain (ASCOBANS, 2012; Viquerat *et al*., 2014).

Major threats include bycatches, pollutants and noise.

7.3.1 References

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7.4 ToR f (d) Azorean ecoregion

In the Azores archipelago, a total of 28 species of marine mammals has been recorded. Marine mammal presence is limited almost exclusively to cetaceans since only a few individuals of harbour (*Phoca vitulina*), grey (*Halichoerus grypus*), harp (*Pagophilus groenlandicus*) and hooded (*Cystophora cristata*) seals have been recorded since the disappearance of the monk seal from the archipelago (Silva *et al*., 2009; SRMCT, 2014). Not all the cetacean species recorded are common and even those found regularly in the area are part of populations whose range extends beyond the region. The most common sighted species include bottlenose dolphin (*Tursiops truncatus*), Atlantic spotted dolphin (*Stenella frontalis*), Risso´s dolphin (*Grampus griseus*), striped dolphin (*Stenellacoeruleo alba*), and sperm whale (*Physetermacro cephalus*). In addition, the Azores are visited in spring by several baleen whales species (blue whale *Balaenoptera musculus*, fin whale *B. physalus*, sei whale *B. borealis* and humpback whale *Megapterano vaeangliae) en route* to their feeding grounds in northern latitudes (Visser *et al*., 2011a; Silva *et al*., 2013; SRMCT, 2014) (see Figure 7.5). Beaked whales (Cuvier's *Ziphius cavirostris* and *Mesoplodon* species, especially Sowerby's beaked whale *M. bidens*) appear to be common and present all year round.

Abundance estimates for these species are available for some island groups of but there are no abundance estimates for the whole archipelago.

It has been suggested that the main anthropogenic threats facing cetaceans while in Azores waters are anthropogenic noise (especially from seismic surveys associated with geophysical research and mining exploration, which represent a particular concern for vulnerable species such as the sperm whale and beaked whales), whale watching, marine litter, other types of pollution, collisions and climate change.

Using photo-identification techniques, some groups of bottlenose dolphins and Risso´s dolphins have been found to be island-associated, showing varying degrees of site fidelity (Silva *et al*., 2008; 2009; 2012; Hartman *et al*., 2014; 2015) and thus at greater risk of disruption by whale watching operations (Visser *et al*., 2011b, Silva *et al*., 2012) compared to other groups.

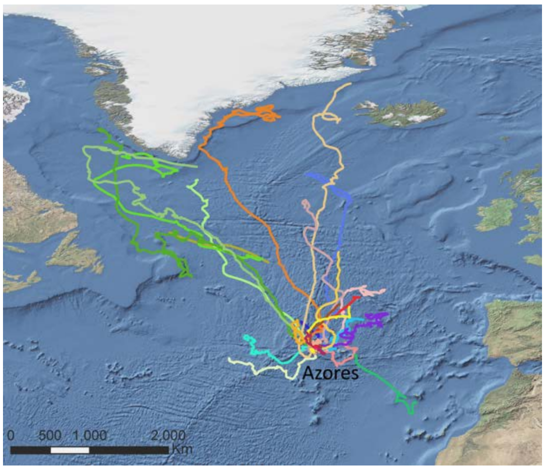

Figure 7.5. Satellite tracks of sei whales (greenish colours), fin whales (pink, orange) and blue whales (blue, magenta and white) instrumented with satellite tags in the Azores between 2008 and 2015 (Silva *et al.*, 2013; Prieto *et al.*, 2014).

7.4.1 References

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7.5 ToR f (e) Oceanic Northeast Atlantic ecoregion

Marine mammals inhabiting these areas are oceanic species that range widely and the distribution of which extends beyond this ecoregion. Several of the largest species (notably the sperm whale and many baleen whale species) were historically hunted for their blubber and meat as part of a worldwide industry that quickly developed over the 18th and 19th centuries. As a consequence of the overexploitation by the whaling industry, many whale populations were seriously depleted. Commercial whaling stopped (due to the IWC "Moratorium"), for all whale species and populations, from the 1985/1986 season. Most whale populations have since experienced a recovery but not all have done so. Thus, blue whales remain rare in the NE Atlantic although they are indications that numbers are increasing, while for sei whales there are no clear signs of recovery.

There are some population abundance estimates for the Northeast Atlantic but not all areas of the ecoregion have been surveyed and many species have not been fully assessed in the area.

A small northern portion of this ecoregion was covered by the NASS (North Atlantic Sightings) surveys (NAMCO, 2016) in 1987, 1989, 1995, 2001, 2007 and 2015 (Figure 7.6). These internationally coordinated surveys were designed to obtain information on cetacean distribution and abundance in the northern North Atlantic.

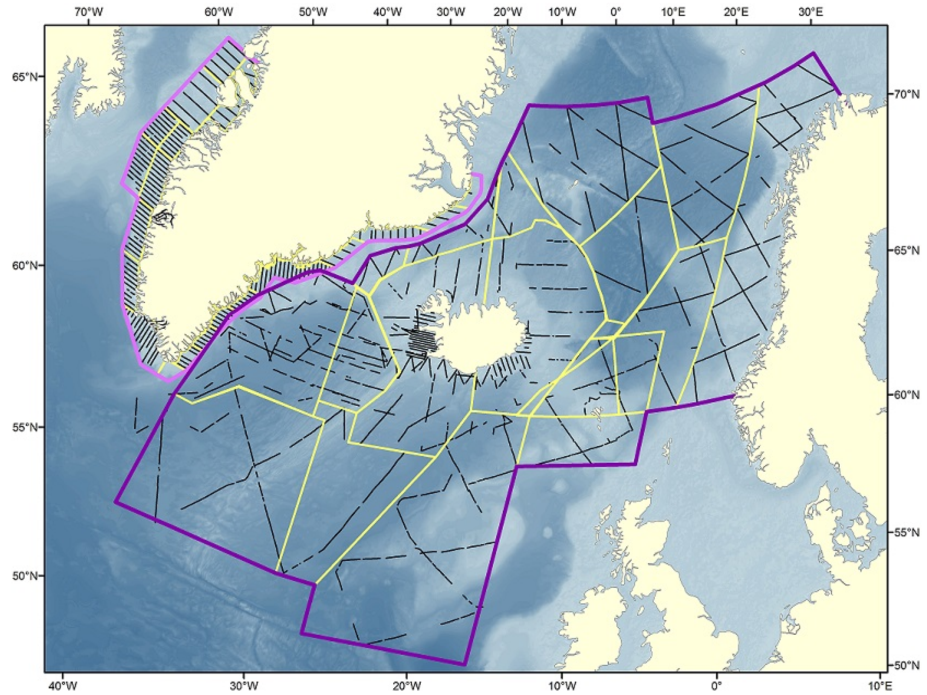

Figure 7.6. Map of the areas surveyed as part of the NASS 2015 survey. Black lines are the survey tracts. Reproduced from NAMCO webpage (www.namco.no).

Other surveys have provided additional data on marine mammal presence and distribution in parts of the region. The MAR-ECO expedition on board the RV "G.O. Sars" in summer 2004 covered the area of the Mid-Atlantic Ridge between Iceland and the Azores. During this expedition eleven cetacean species were identified, with sei and sperm whales been the most commonly sighted species. The most important area for baleen and sperm whale sightings was the Charlie Gibbs Fracture Zone, although sperm whales were also observed north of this area. More pilot whales (*Globicephala* sp.) and Atlantic white-sided dolphins (*Lagenorhynchus acutus*) were observed in the cooler and less saline waters while the opposite was true for common and striped dolphins (Waring *et al*., 2008).

An expedition organised by the IFAW on board the Song of the Whale in 2012 reported marine mammal presence based on acoustic and visual detections during four passages between the Azores, the Gulf of Maine and Iceland that took place in summer and autumn. Sixteen species of cetaceans were detected over the 66 days on effort. Common dolphin was the most commonly encountered species. The survey also detected harbour porpoises acoustically over the Mid-Atlantic and Iceland–Faroe Ridges, in waters up to 2000 m deep (Ryan *et al*., 2013).

Since 2012, the CETUS project has collected distribution data in the southern part of this ecoregion, using marine mammal observers based on commercial cargo ships covering the routes between mainland Portugal and Madeira, Azores, Canary and Cape Verde islands. To date 87 trips have been completed. These routes cross the ecoregion between the latitude 36ºN and 40ºN during summer months (July–October). A total of 181 sightings has been recorded and ten species were identified, including various dolphins (113 sightings), toothed whales (29 sightings) and baleen whales (31 sightings). Striped and spotted dolphins (*Stenella coeruleoalba* and *S. frontalis*) were the species most frequently sighted and with the biggest group size recorded (Figure 7.7).Preliminary results appear in Correia *et al*. (2015).

Threats to marine mammals in these areas include bycatch, pollution, climate change, ship traffic.

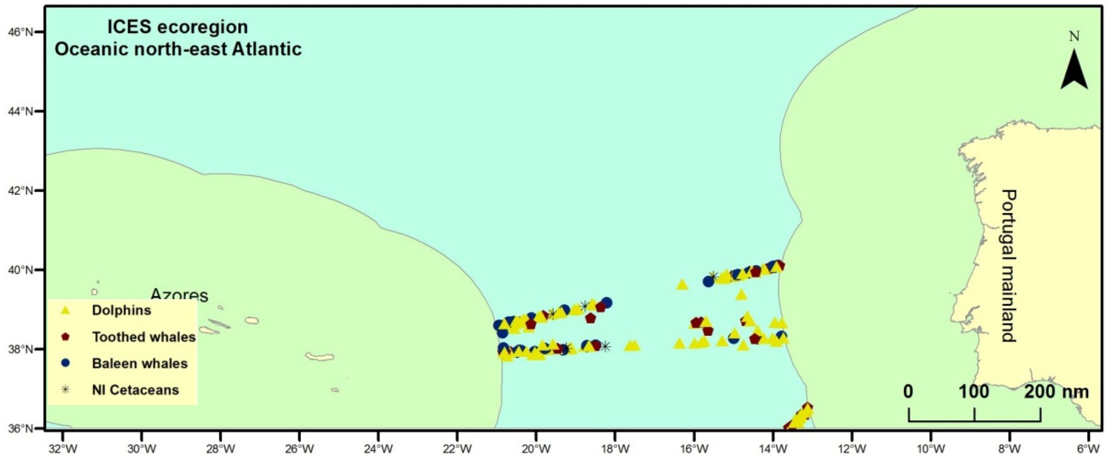

Figure 7.7. Distribution of cetacean sightings recorded within the oceanic North Atlantic region during monitored trips on commercial cargo ships (CETUS project).

7.5.1 References

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Annex 2: Recommendations

None.

Annex 3: Draft Terms of Reference 2018

WGMME – The Working Group on Marine Mammal Ecology

2017/X/ACOMXX The **Working Group on Marine Mammal Ecology** (WGMME), chaired by Anders Galatius (Denmark) and Anita Gilles (Germany), will meet in La Rochelle, France, X–X February 2018 to:

a) Review and report on any new information on seal and cetacean population abundance, population/stock structure, management frameworks (including indicators and targets for MSFD assessments), and anthropogenic threats to individual health and population status;

b) Review current issues in relation to indirect impacts of seals on fisheries;

c) Review additional aspects of marine mammal fishery interactions not covered by WBYC. Details of this ToR to be agreed with WGBYC;

d) Update the database for seals;

Justification

ToR a is a standing term of reference. However, the group proposes to expand its scope since it would be useful to include information on threats to population status.

ToR b aims to address current issues in indirect seal–fisheries interactions (e.g. competition for food, transmission of codworm), complementing the review of direct interactions completed in 2017.

ToR c is proposed in the recognition of common interests between WGMME and WGBYC, recognising that some issues related to marine mammal-fishery interactions may finally be covered by neither group.

ToR d is a standing term of reference.

Annex 4: Working document

See below.

HERO ON BOARD!!!!!

BY HURRICANE FORCE 5

HURRICANE FORCE 5

- Danny, Gayatri, Julia, Lauren, William
- Team Managers: Brian Horton and Vishnu Srinivasan *(taking the picture!)*
- Mentors: Joseph Asfouri and Leo Orshansky

PROBLEM-BYCATCH

- The definition bycatch is the accidental capture of non targeted cetacean and other marine life
- Marine animals are getting caught in fishing nets and getting killed and injured

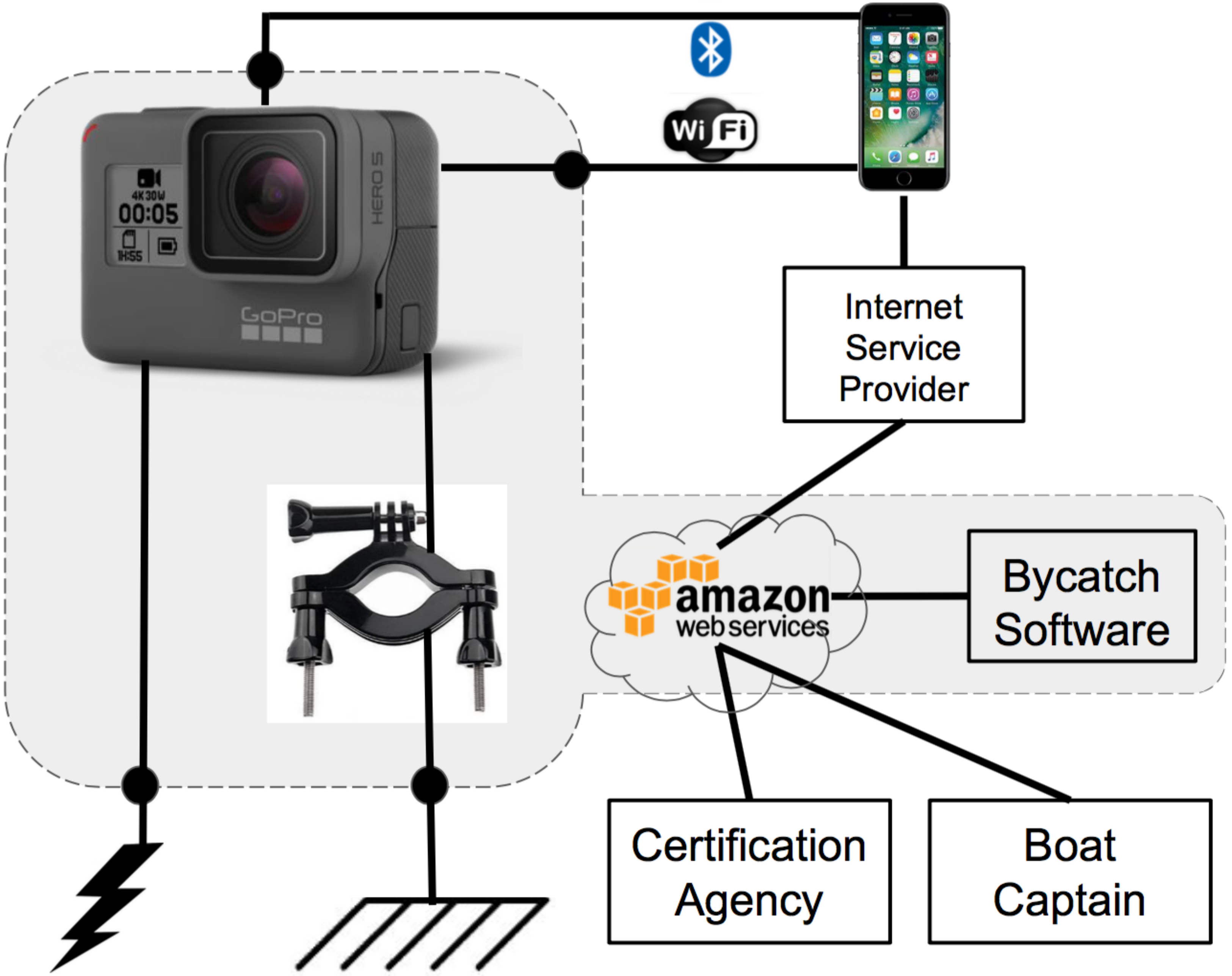
WHY DOES THE PROBLEM STILL EXIST

- 298,000 turtles are killed annually!
- 300,000+ dolphins are killed each year!

- Current solutions that are not effective
  - Pingers
  - Observers and monitors on boats
  - Fishing Rules and Regulations

OUR SOLUTION

Our solution is to have a camera on the boat surveying the fishing nets and taking pictures every 20 seconds. It will then send pictures to a computer so image recognition software can look for bycatch. If there is no bycatch, the fish will be certified as Dolphin Safe or Sea Turtle Safe.


Wi Fi
Internet Service Provider
amazon web services
Bycatch Software
Certification Agency
Boat Captain

"HERO" SESSION BLACK 5 ON BOARD

- GPS
- Burst time lapse
- Waterproof
- Wifi, Bluetooth enabled
- Advanced wind reduction
- Auto upload to iCloud
- Video stabilization

THANK YOU

- **Questions**