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

19–22 February 2018

La Rochelle, France

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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Executive summary

The Working Group on Marine Mammal Ecology (WGMME), chaired by Anita Gilles (Germany) and Anders Galatius (Denmark), met in La Rochelle (France), 19–22 February 2018, to work on four terms of reference (ToRs).

Two of these were standing terms of reference; under the first of these, ToR A, new and updated information on seal and cetacean population abundance, population/stock structure, management frameworks, and anthropogenic threats to individual health and population status were reviewed. The group expanded the scope for ToR A, including reviews of population trends of seal stocks in the Baltic Sea and Wadden Sea, as well as producing charts illustrating population trends of seals in the North Atlantic, where data could be made available. For cetaceans, information is provided regarding the passive acoustic monitoring of harbour porpoises in the Baltic Sea as well as updates regarding visual survey monitoring and strandings of several cetacean species. With respect to the development of common indicators and targets for the Marine Strategy Framework Directive, updates from France and the Macaronesian region are provided. A revision of the delineation of assessment units for harbour porpoises in the Belt Sea is discussed. New information on anthropogenic stressors have been compiled and a further stressor category "Tourism" has been introduced.

Under ToR B, WGMME reviewed current issues in relation to indirect impacts of seals on fisheries (direct impacts were reviewed in the 2017 report). The review includes a coverage of competition for resources (fish stocks), also reviewing the latest information on seal diet, and the role of seals in the transmission of nematode parasites.

ToR C was originally implemented to review aspects of marine mammal fishery interactions not covered by ICES WBYC. However, it was not possible to obtain information on the topics to be covered by WGBYC in time before the WGMME meeting. WGMME therefore decided to produce a review of recent marine mammal bycatch data and development of mitigation measures.

ToR D, updating the database for seals, is also a standing term of reference. WGMME decided to thoroughly rework the ICES WGMME SEAL database to provide a useful source for the standing ToR A. WGMME also endorsed the format for a data call proposed by OSPAR to provide data for assessments under OSPAR indicators M3 and M5 on seal abundance and distribution. WGMME found the proposed data submission format relevant and useful and will assist OSPAR in the data call.

1 Introduction

The Working Group on Marine Mammal Ecology (WGMME) met at the Institut du Littoral et de l'Environnement, University of La Rochelle in France, during 19–22 February 2018. 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 of La Rochelle, especially Cécile Vincent and Florence Caurant, for hosting the meeting.

The Chairs 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.

The Working Group gratefully acknowledges the support given by several additional experts that kindly provided information and/or reports for use by WGMME. These included Olli Loisa (Turku University of Applied Sciences, Finland), Michal Malinga (DHI Polski, Poland), Anja Gallus (German Oceanographic Museum, Germany), Oliver Ó Cadhla (Ireland), Genevieve Desportes (NAMMCO), Sara Königson (Swedish University of Agricultural Sciences), Finn Larsen (DTU Aqua, Denmark) and Jan Lakemeyer (University of Veterinary Medicine Hannover, Foundation).

The WGMME updated ToRs for 2019 (see Annex 2) and discussed meeting venues. The University of Veterinary Medicine Hannover, Foundation, offered to host the 2019 meeting in the Institute for Terrestrial and Aquatic Wildlife Research in Büsum (Germany).

2 ToR 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

New information on seal and cetacean abundance, including distribution, and population/stock structure, as well as management frameworks and anthropogenic threats is reviewed and reported below. New information on fisheries bycatch is included under ToR (c).

2.1 New abundance information

2.1.1 Seals

Table 1, Table 2 and Table 3 summarise the most recent available seal survey data, analogous to what WGMME has presented in former years. In the following, a thorough assessment of population stocks is presented individually for the different countries/management units and species, including trajectories of (available) counts.

Unless it is stated that a figure refers to a population abundance estimate, numbers of seals reported are those counted on haul-outs which do not include seals at sea during surveys.

Table 1. Recent harbour seal survey data.

Country		Survey Year(s)	Adults (moult)	Pups	References
Norway					Nilssen and Bjørge 2017 a & b
	North of 62N	2015	3872		
	South of 62N	2011–2016	1128		
	Finmark	2012–2013	981		
	Skagerrak	2015–2016	638		
Iceland		2016	7652		Thorbjörnsson *et al.*, 2017
Wadden Sea		2017	26000	9167	Galatius *et al.*, 2017
Dutch Delta Area		2016/2017	685 (2016)	50 (2017)	Arts *et al.*, 2017
France		2017	1083	179	Vincent *et al.* (in revision)
UK					
	Scotland	2011–2016	25149		SCOS, 2017
	England and Wales	2015–2016	5185		SCOS, 2017
	Northern Ireland	2011	948		SCOS, 2017
Ireland		2011–2012	3489		Duck and Morris, 2013
USA		2012	75834		Waring *et al.*, 2015
Canada					NAMMCO
	south of Labrador	1970s	12700		
	Estuary and Gulf of St Lawrence	1994–2000	4000–5000		
Sweden and Denmark					
	Skagerrak	2016	6577		Swedish Museum of Nat. Hist., Markus Ahola
	Kattegat/ Danish Straits	2016	10546		Swedish Museum of Nat. Hist., Markus Ahola or HELCOM
	southern Baltic	2017	974		HELCOM
	Limfjord	2016	1097	467	HELCOM
	Kalmarsund	2016	1100		HELCOM

Table 2. Recent grey seal survey data.

Country		Recent Survey Year(s)	Pups	Adults (moult)	References
Norway	*Tomso & Finmark*	2015–2016	271		Nilssen and Bjørge, 2017a & b
	Norway north of 62N	2014–2015	318		Nilssen and Bjørge, 2017a & b
	Norway south of 62N	2017	40		Nilssen and Bjørge, 2017a & b
Iceland		2012	992	4.200	Hauksson *et al.*, 2012
Wadden Sea		2017	1279	5.445	Brasseur *et al.*, 2017
Dutch Delta Area		2017	2*	1.358	Arts *et al.*, 2017; *pups born elsewhere (UK) can strand in these areas
France		2016	43	895	Vincent *et al.* (in revision)
UK	*Inner Hebrides*	2014	4054		SCOS, 2017
	Outer Hebrides	2014	14348		SCOS, 2017
	Scottish North Sea	2014, 2004*	32842		SCOS, 2017; * Shetland
	English North Sea	2016	8157		National Trust, Lincolnshire Wildlife Trust, Natural England, Friends of Horsey Seals
	SW England & Wales	2005	1900		SCOS, 2017
Republic of Ireland		2012	2100		Ó Cadhla *et al.*, 2013
Canada	*Sable Island*	2016	83 594		den Heyer *et al.*, 2017
	Gulf of St Lawrence + eastern shore Canada	2016	15 090		den Heyer, *et al.*, 2017; Hammill *et al.*, 2017
USA	*USA east coast*	2013	3037		http://www.nefsc.noaa.gov/publications/tm/tm238/247_f2015_grayseal.pdf
Baltic	Baltic	2017		30 000	HELCOM

Table 3. Recent ringed seal survey data.

Country		Survey Year(s)	Adults (moult)	Pups	References
Sweden, Finland	*Bothnian Bay*	2017	13 664		HELCOM (normal ice conditions)
	Bothnian Bay	2015	19 936		HELCOM (unusual ice conditions)
Estonia, Finland, Russia	*Gulf of Finland*	2017	80		Vervkin and Voyta, 2017 (average taken of range)
Estonia, Latvia	*Gulf of Riga*	2013	1526		M. Jussi, pers. comm., 2013
Finland	*Finnish Archipelago Sea*	2017	200–300		Nordström *et al.*, 2011; Halkka and Tolvanen, 2017

ICELAND: Icelandic harbour seal (*Phoca vitulina*) and grey seal (*Halichoerus grypus*) populations are currently in decline. The harbour seal population has 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 (Thorbjörnsson *et al.*, 2017). The Icelandic grey seal population has been surveyed at irregular intervals since 1982 when the population abundance was estimated at 9000 animals. The latest estimate from 2012 indicated a population abundance of 4200 animals (Hauksson *et al.*, 2014). A new grey seal census was carried out in 2017 and analysis is underway.

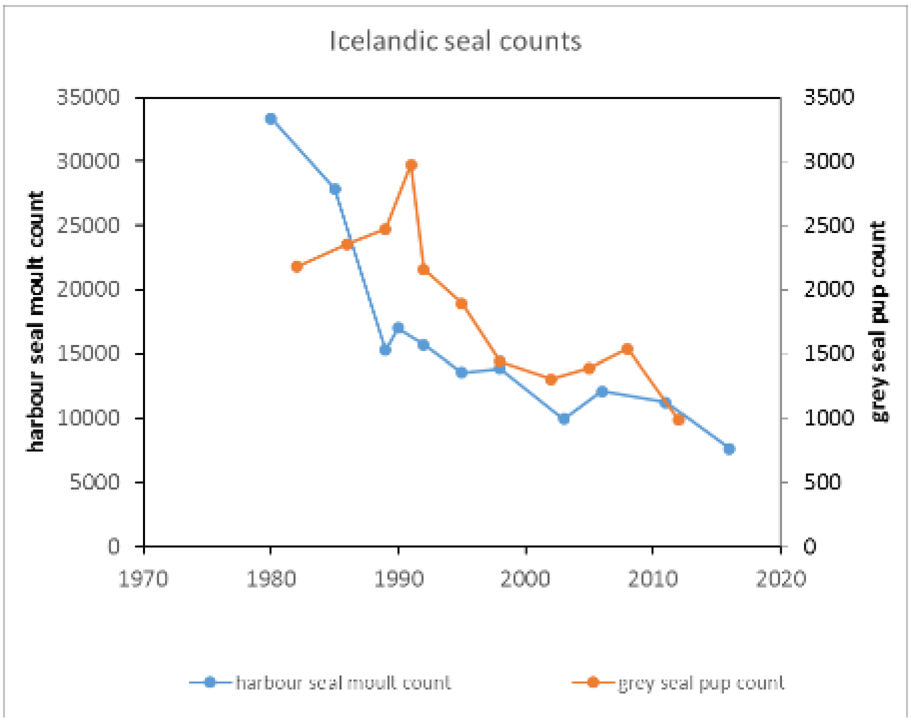

Figure 1. Trends of counts of moulting harbour seals and grey seal pups in Iceland.

2.1.1.1 Baltic Sea

Ringed seals

Ringed seal (*Phoca hispida* ssp. *botnica*) breeding and moulting distribution is connected to sea ice in winter and spring. Since ringed seals haul out scattered on ice during their annual moult, they have traditionally been surveyed using line-transect methodology. Favourable ice-conditions usually occur every year in the Bothnian Bay, where the surveys have been carried out since 1988. The number of hauled out individuals during the surveys has increased from the level of 2000 to 8000, corresponding to an annual average population increase being 4.5% per year. However, after recent exceptionally warm winters, the sea ice has started to break before or during the peak of moulting and survey time, leading to potentially anomalous survey results of between 10 000 and 20 000 individuals in 2013, 2014, 2015 and 2017. These results have been excluded from the trend analysis as they are not considered comparable to previous data. However, they do reveal that the proportion of ringed seals hauling out during the surveys on fast ice is smaller than previously thought and that the true population abundance size exceeds 20 000 animals in the Bothnian Bay. Further research and relevant quantitative measures for the ice quality are needed for better understanding of the haul-out behaviour of Baltic ringed seals in order to calibrate the results in different ice-conditions and to estimate the true population size. Apart from challenges for population monitoring, warming climate supposedly leads to negative impacts on ringed seal populations. The extent of sea ice and snow-cover that shelters pups from harsh weather and predators is diminishing, degrading the breeding habitat and lowering the reproductive success of ringed seals.

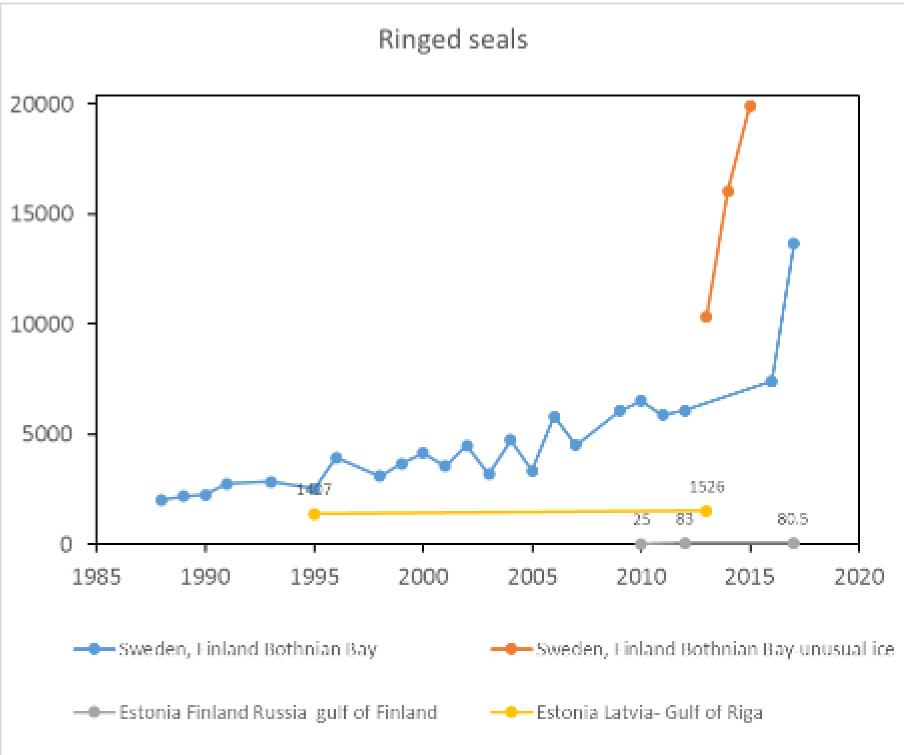

Figure 2. Trends of estimates of numbers of ringed seals hauled out on sea ice during moult in the Baltic.

***Southern ringed seal populations in the Baltic Sea*:** As a result of population decline during the 20th century, the current ringed seal population is divided into four geographical subpopulations: Next to the larger sub population in the Bothnian Bay, subpopulations can be found in the Gulf of Riga, the Finnish Archipelago Sea and the Gulf of Finland. There is no evidence of exchange of individuals between the subpopulations from telemetric studies (Härkönen *et al*., 1998; M. Jüssi, unpubl.) although long-distance movements outside the breeding season occur (Oksanen *et al*., 2015). While the warmer winters challenge population monitoring of ringed seals in the Bothnian Bay, traditional surveys have been impossible in the areas occupied by the southern subpopulations in most years. The lack of continuous monitoring data provides a severely fragmented view of population development, although the existing survey results indicate stable or decreasing trends. The subpopulations of Gulf of Riga and Finnish Archipelago Sea have been estimated to consist of about 1000 individuals (M. Jüssi, pers. comm, 2013) and 200–300 (Nordström *et al*., 2011; Halkka and Tolvanen, 2017), respectively. In the southern areas, counts from land haul-outs during the ice-free period have been developed and carried out. These results do not show notable trends relative to the last estimates based on ice distribution.

The fourth subpopulation, inhabiting the Gulf of Finland, has suffered a population collapse from an estimated 3500 individuals in 1980s to less than 200 individuals in 1995 (Härkönen *et al*., 1998). Counts on haul-outs indicate an easterly contraction of the population distribution and a further drop in abundance. Aerial surveys in 2010–2012 and 2017 yield abundance estimates of less than 100 individuals (upper 95% CI limit) hauling out on spring ice (M. Verevkin, unpubl.). Reasons for the population decrease remain unknown. Movements of seals marked with telemetry tags indicate that the current distribution is confined to island and reef systems in the easternmost part of the gulf (east of 25°E). Due to mild winters, stable ice cover forms only in the north-eastern part of the gulf (Vyborg Bay area); this area was used for breeding by the tagged seals (M. Jüssi, M. Verevkin unpubl.).

Reduced extent and duration of sea ice with less snow compared to historically average winters, probably decreases breeding success of the population through early loss of stable ice as breeding platform and exposure of pups to predators and anthropogenic pressure factors as well as lack of snow for lairs. Thus, there has been a significant loss in a key habitat extent and quality caused by climate change. Apart from the warming climate, the anthropogenic pressure in the area is potentially high in form of shipping, ice breaking, fishing and marine pollutants (Raateoja and Setälä, 2016). Combined with the low population numbers, very limited distribution range and the unfavourable climatic conditions for this ice-dependent pinniped, the population recovery perspective for the Gulf of Finland subpopulation is unfavourable.

Baltic ringed seals have been classified as 'Vulnerable' under the HELCOM Red List (2013) and under a previous IUCN assessment (2009), but as 'Least Concern' in the latest IUCN assessment (2015; see Härkönen, 2015). Despite these classifications, the threats of climate warming apply to all southern subpopulations of the Baltic subspecies, which are facing a risk of regional extinction. Depopulation of the southern remains of the historical breeding distribution will lead to a significant loss of the total breeding range of the Baltic ringed seal. Although the species is recovering from the lowered reproductive ability caused by environmental contaminants and the subpopulation in the Bothnian Bay is currently growing, albeit at a low rate, the projected negative trends in suitable breeding habitat and reproductive success due to climate warming are threatening the whole subspecies in a longer perspective.

Grey seals

Monitoring of grey seal population in the Baltic Sea is based on internationally coordinated censuses during the moulting season, covering the entire Baltic moulting distribution of the species. The maximum number (not corrected for individuals in water) counted during 2–3 replicate surveys in each sea area is used for assessing population development. The grey seal population in the Baltic has been growing throughout the span of the coordinated surveys (starting in 2003) with the most pronounced growth in the southern and western parts of the moulting distribution. During recent years, however, the growth has shown signs of stabilising, which can be an indication of approaching carrying capacity of the current Baltic Sea environment. The counted number has been at the level of 30 000 animals during recent years (HELCOM, unpubl.). Of the hauled-out population, about 80% is found in the core moulting area around the central Baltic proper (archipelagos of Central Sweden, southwestern Finland and Western Estonia). Outside the breeding and moulting seasons, grey seals travel and forage in other areas, too.

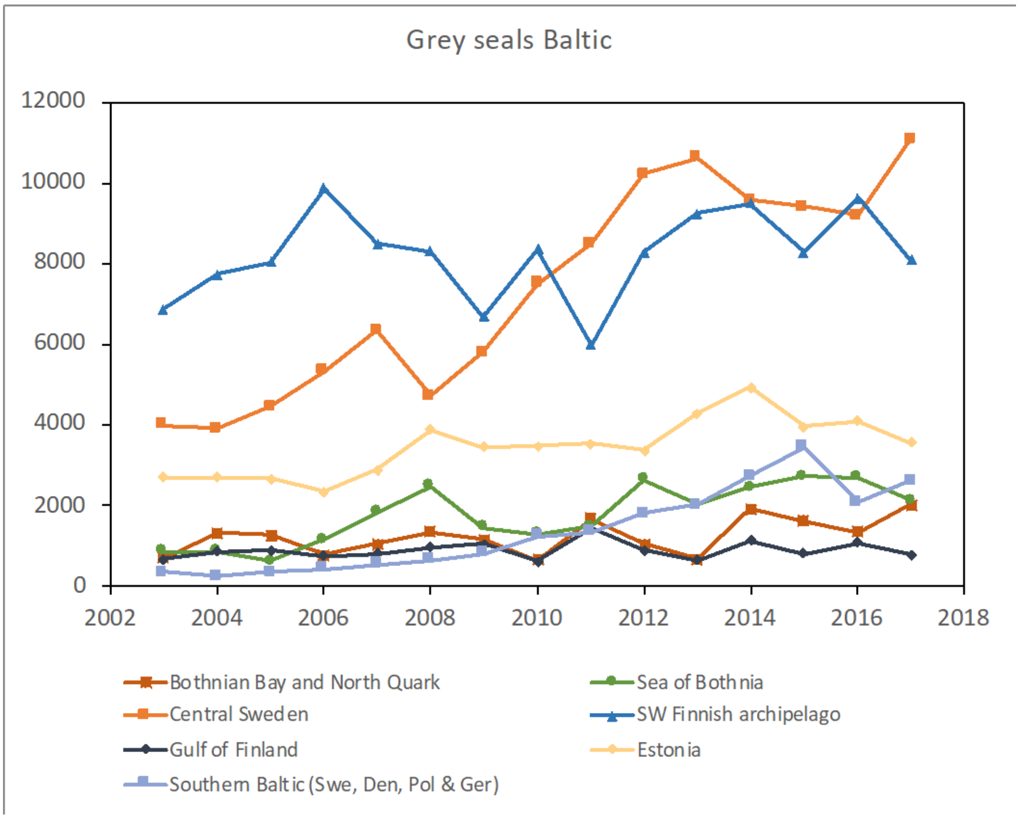

Figure 3. Trends of counts of moulting grey seal in subareas of the Baltic Sea.

Harbour seals

Harbour seals in the Baltic (HELCOM) area (Denmark and Sweden) are surveyed annually using replicate annual aerial surveys during the moulting period in August. They are split into the four management units: Limfjord, Kattegat, Southwestern Baltic and Baltic Proper (Kalmarsund).

LIMFJORD: The number of counted seals of the Limfjord harbour seal population has been fluctuating around 1000 individuals since the early 1990s and appears to have reached its carrying capacity. Genetic analyses indicate that the seals in the fjord originate in two different populations, (1) the population originally inhabiting the fjord, before a storm opened the passage to the North Sea in 1825, and (2) seals from the Wadden Sea (Olsen *et al*., 2014). It is not known to what extent the seals from the Wadden Sea use the fjord for other purposes than hauling out and to which extent they interbreed with the native seal population. A proper assessment of the Limfjord harbour seals is contingent on clarification of these issues. In 2016, 900 seals were counted in the fjord (HELCOM, 2017).

KATTEGAT: The harbour seal population in Kattegat and the northern Danish Belt Sea experienced two dramatic mass mortality events due to PDV when more than 50% of the population died in 1988 and about 30% in 2002 (Härkönen *et al*., 2006). Unusually large numbers also died in 2007, but the reason for this mortality remains unclear (Härkönen *et al*., 2007). In the spring and summer of 2014, some seals appearing to show signs of pneumonia were found in Sweden and Denmark. Avian influenza H10N7 were isolated from a number of these seals (Zohari *et al*., 2014; Krog *et al*., 2015; Bodewes *et al*., 2016). The rate of increase between the two PDV epidemics was close to 12% per year as in the adjacent North Sea populations. The annual population growth rate in Kattegat and the Danish Belt Sea remained close to 12% per year until 2010, but data suggest that it is levelling off, even if the increased mortality in 2014 is taken into account. This is likely to be caused by density-dependence, indicating that the population is approaching carrying capacity. Counted number was 9400 in 2016 (HELCOM, 2017).

SOUTHWESTERN BALTIC: Southwestern Baltic harbour seals were also hit hard by the PDV epidemics of 1988 and 2002. Since the 2002 epidemic, the population has grown with an average annual rate of 6.4%, with indications of a declining trend in recent years. In 2016, 1000 seals were counted in the area (HELCOM, 2017).

BALTIC PROPER/KALMARSUND: The harbour seal population in Kalmarsund is genetically divergent from adjacent harbour seal populations (Goodman *et al*., 1998) and experienced a severe bottleneck in the 1970s when only some 30 seals were counted. Long-term isolation and small numbers have resulted in low genetic variation in this population (Härkönen *et al*., 2006). The population has increased annually by ca. 9% since 1975 and counted numbers amounted to about 1100 seals in 2016. (HELCOM, 2017).

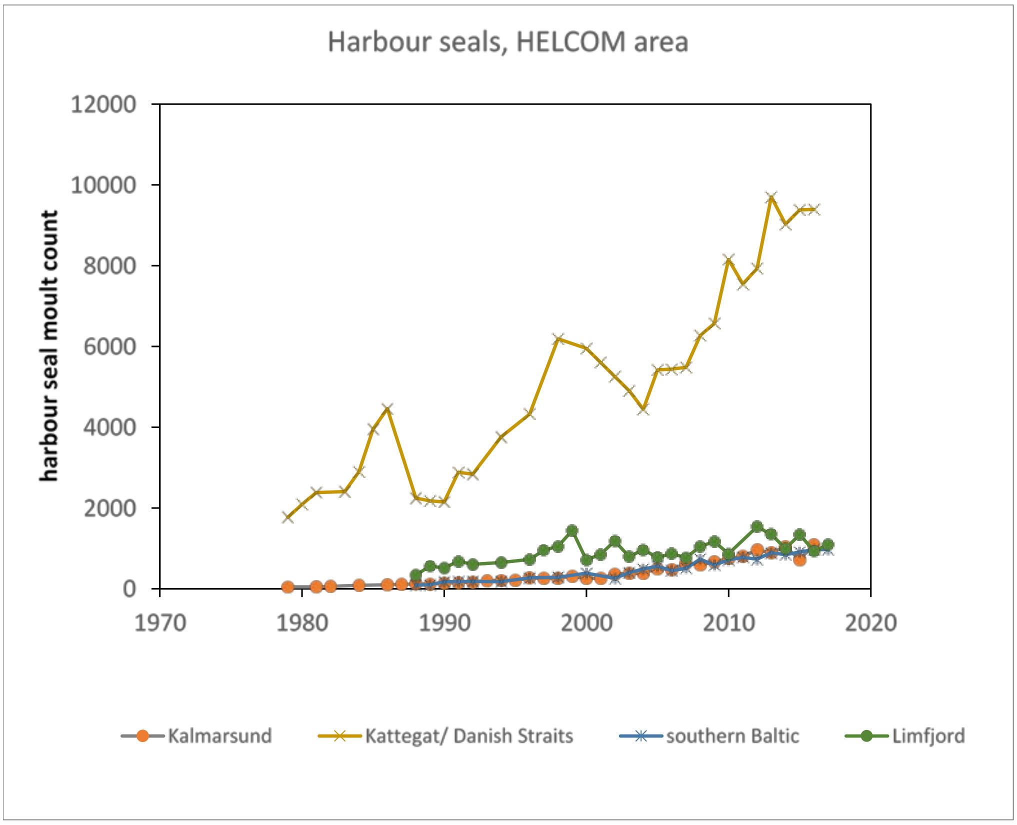

Figure 4. Trends of moult counts of harbour seals in the Kattegat, Southwestern Baltic, Limfjord and Kalmarsund.

2.1.1.2 Skagerrak

Harbour seals

The Skagerrak harbour seal population collapsed by roughly 50% during two mass mortality events due to PDV parallel with the Kattegat population in 1988 and 2002. Before the two collapses, the population increased with high rates indicating no factors retarding the growth. After the later collapse, the rate of increase has been lower which may indicate approaching carrying capacity. The counted number of harbour seals in Skagerrak was at the level of 6500 in 2016 (not corrected for seals at sea during the surveys).

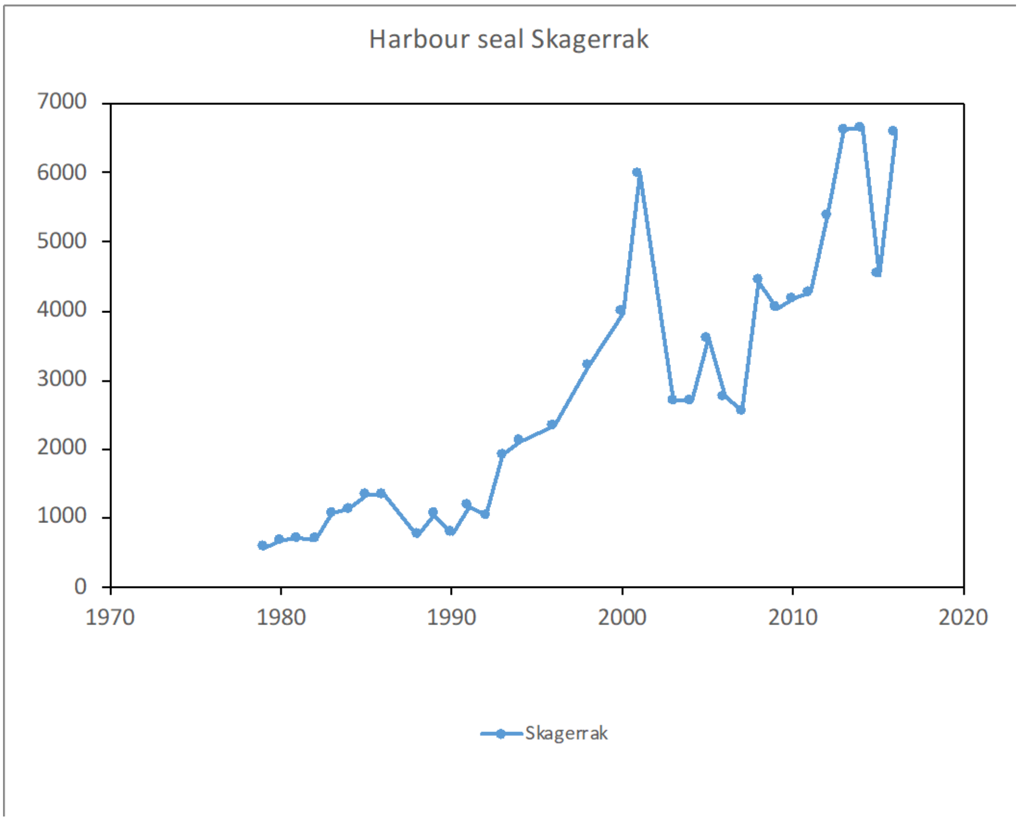

Figure 5. Trends of moult counts of harbour seals in the Skagerrak.

2.1.1.3 Continental coast, Wadden Sea to France

Harbour seals

Harbour seal surveys in the Wadden Sea are coordinated among Danish, German and Dutch scientists. Brasseur *et al.* (2018) investigated a 40-year time-series (1974–2014) of counts of harbour seals in the Wadden Sea to study underlying processes of recovery and demonstrated the influence of historical regional differences in management regimes on the recovery of this population. Mortality rates were close to 50% during both PDV epidemics in 1988 and 2002, and between and after the epidemics, population growth rate has been close to the maximum intrinsic exponential growth rate of harbour seals at 12–13%. During recent years, growth in moult counts has levelled off, although pup counts continue to increase. In 2017, almost 26 000 harbour seals were counted during the moult (not corrected for seals at sea during the surveys) (Galatius *et al.*, 2017).

SOUTHERN NETHERLANDS, BELGIUM and FRANCE. The growing seal colony in the Dutch Delta area in the southern Netherlands is thought to be a colony of the Wadden Sea population as there are not enough local births to explain this growth and telemetry shows regular exchange between the areas. Approximately 700 animals were counted in the Dutch Delta area in 2015 (Arts *et al.*, 2017), and numbers have been growing at almost 15% annually since 2002. A similar exchange might occur with the French colonies though here local births and exchange with southern English colonies might also play an important role in the growth. In 2017, seal counts amounted to almost 1100 seals in the colonies on the coasts of Brittany and Normandy (Vincent *et al.*

(in revision); Parc naturel marin d'Iroise, ONCFS, Réserve naturelle des sept iles, Bretagne Vivante, Syndicat Mixte Baie du Mont-Saint-Michel, Réserve naturelle nationale du Domaine de Beauguillot, GMN, Picardie Nature, ADN, GDEAM62, CMNF).

In Belgium there are no true seal colonies, however tens of animals strand annually along the coasts. In 2017, the number of washed ashore dead seals was the highest ever, with a total of 37 animals (ten harbour seals, eight grey seals and 19 specimens not identified to species level). In addition, SEALIFE took care of six grey and 22 harbour seals. At least four harbour seals had sustained injuries due to fishhooks (also see ToRs B & C).

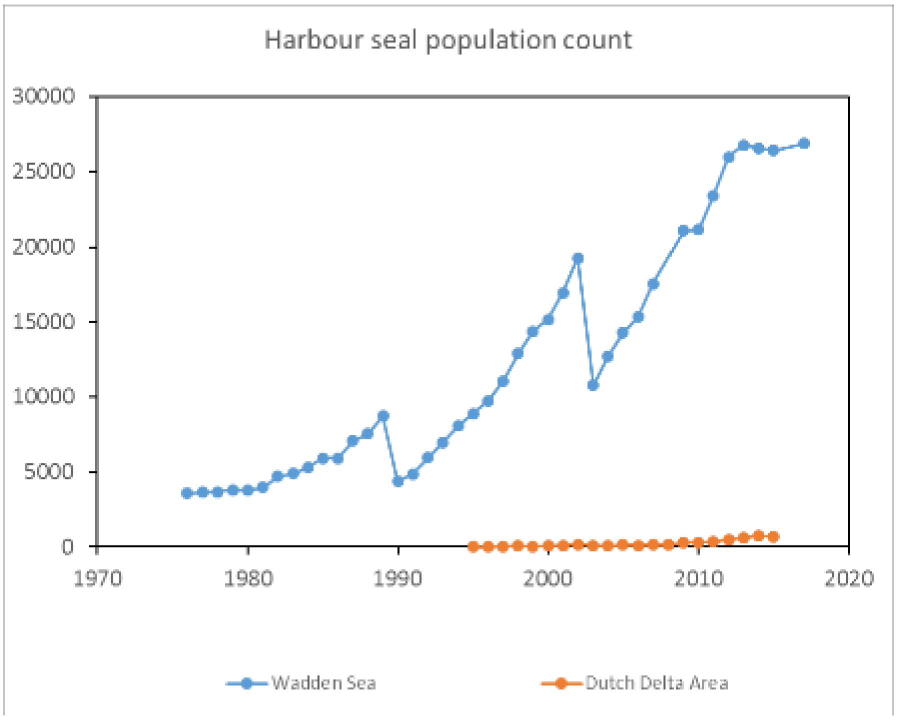

Figure 6. Trends of counts of harbour seals in the Wadden Sea and the Dutch Delta Area.

2.1.1.4 UK

Harbour seal

Table 4. The most recent August counts of harbour seals at haul-out sites in Britain and Ireland by seal management unit, compared with three previous periods: 1996–1997, 2000–2006 & 2007–2009. Details of sources and dates of surveys used in each compiled regional total are given in SCOS-BP 17/03.

Seal Management Unit / Country	Harbour seal counts			
	2011–2016	2007–2009	2000–2006	1996–1997
Scotland Total	25 149	20 430	23 423	29 514
England & Wales Total	5185	4032	3048	3280
Northern Ireland Total	948	1101	1176	
Republic of Ireland Total	3489	2955	2955	
Britain & Ireland Total	34 771	28 518	30 603	

2.1.1.5 Wadden Sea, southern Netherlands, Belgium and France

Grey seals

After centuries of practical absence, grey seals have shown a remarkable recovery in the Wadden Sea area. Partially fuelled by immigration from the UK (Brasseur *et al.*, 2015), colonies started in Germany and the Netherlands and are seen to expand to Denmark. As with harbour seals, grey seal numbers are also growing in the Delta area, despite the complete lack of births. This suggests a continuous exchange between this area and the Wadden Sea and the UK where numbers are growing. In France, there are breeding colonies, although numerous exchanges with the UK and the Wadden Sea have also been recorded with telemetry.

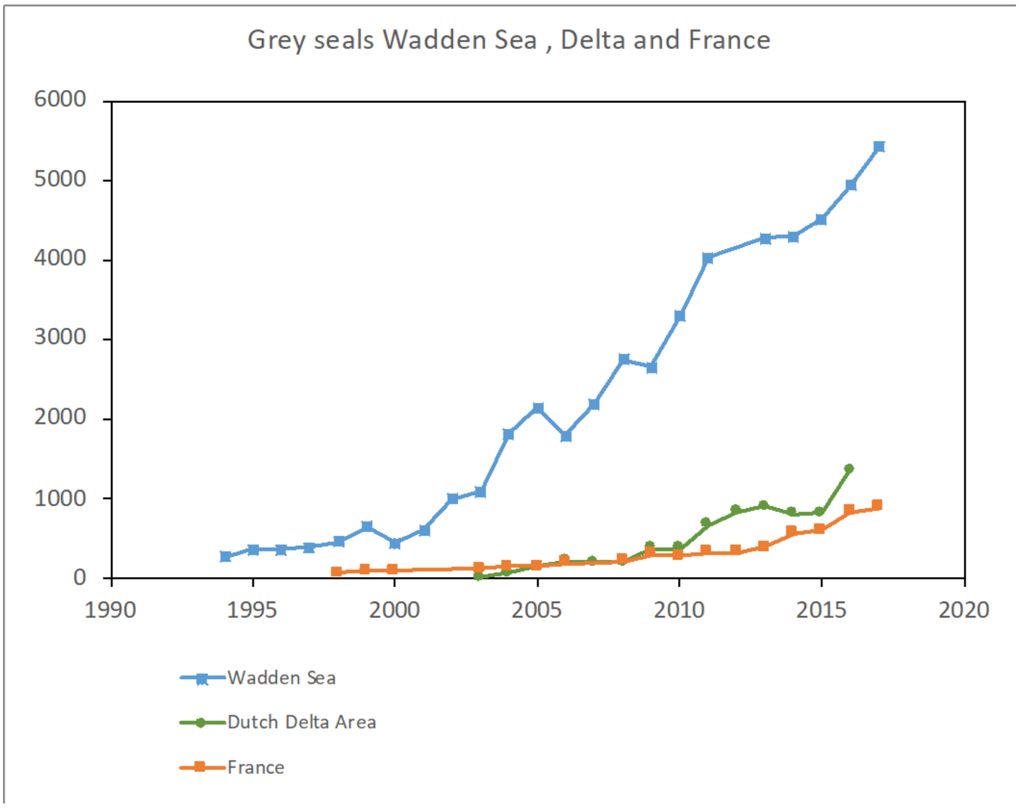

Figure 7. Trends of moult counts of grey seals in the Wadden Sea, Dutch Delta Area and France.

In Great Britain, grey seal population trends are assessed from the counts of pups born during the autumn breeding season, when females congregate on land to give birth. The most recent aerial surveys of the principal Scottish grey seal breeding sites were conducted in 2016. The image processing and counting is not yet available for this report. The most recent results from the 2014 surveys, together with the 2014 estimates from the annually ground counted sites in eastern England, produced a pup production estimate of 54 600. Adding in an additional 5900 pups estimated to have been born at less frequently surveyed colonies in Shetland and Wales as well as other scattered locations throughout Scotland, Northern Ireland and Southwest England, resulted in an estimate of 60 500 (95% CI 53 900–66 900, rounded to the nearest 100) pups (SCOS 2017). Trends of grey seal pup counts from subareas of the UK are shown in Figure 8.

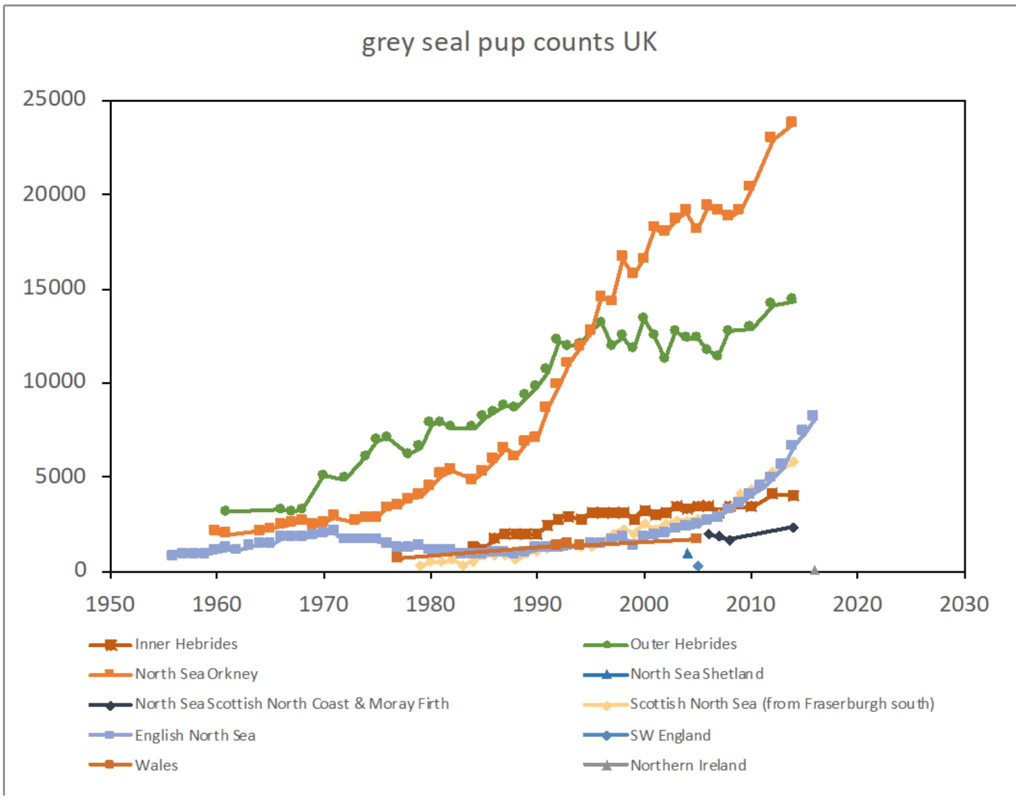

Figure 8. Trends of pup counts of grey seals in subareas of the UK.

Along the North American east coast, grey seal population trends are assessed from the counts of pups born during the breeding season. In Canada, grey seal pup production in 2014 was estimated to be 93 000 pups (95% CI=48 000–137 000), with a total population of 505 000 (95% CI=329 000–682 000). The pup production on Sable Island is estimated to account for about 77% of the estimated total number of pups born in 2014. The estimated 2014 total population of each herd was 394 000 (95% CI 238 000–546 000), 13 800 (95% CI=9300–27 300), and 98 000 (95% CI=54 000–17 ,000), for the Sable, Coastal Nova Scotia and Gulf of St Lawrence herds respectively (Hammill *et al.*, 2014). A smaller, but growing number of grey seal pups are born along the US east coast in Maine and Massachusetts. Trends of grey seal pup counts from the North American east coast are shown in Figure 9.

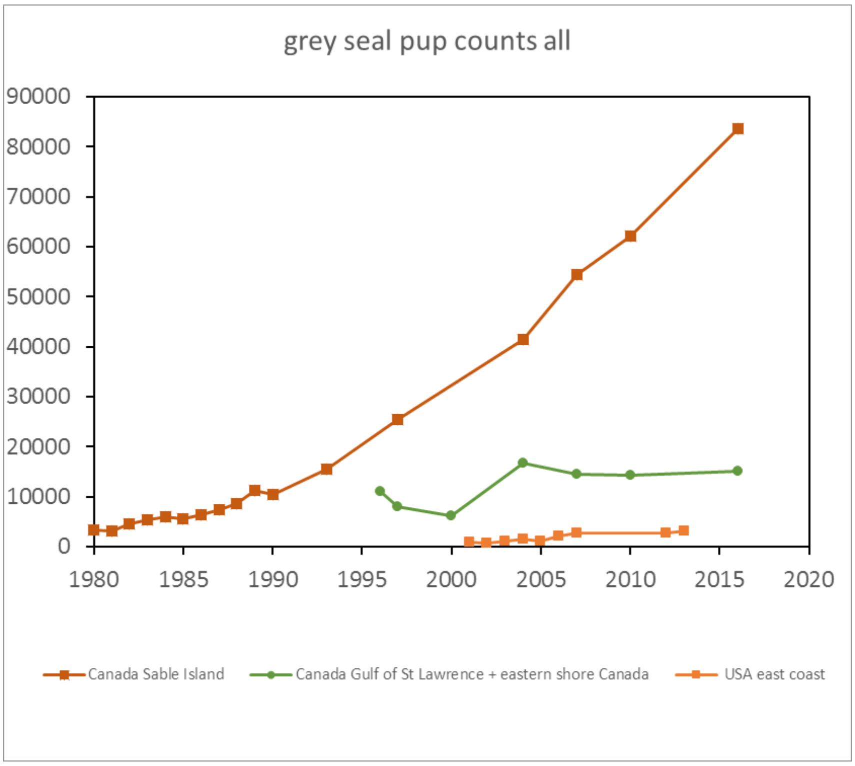

Figure 9. Trends of pup counts of grey seals along the east coast of North America.

2.1.1.6 Rescue and rehabilitation of seals

THE NETHERLANDS. Brasseur (2018) reported on numbers of seals stranded dead and numbers brought into rehabilitation centres based on data from a public database on which all wildlife observations can be placed by any member of the public (www.waarneming.nl). Data are authenticated by a controller before being published.

The number of seals brought into rehabilitation centres and seals found dead relative to numbers of seals counted are shown in Figure 10 and Figure 11.

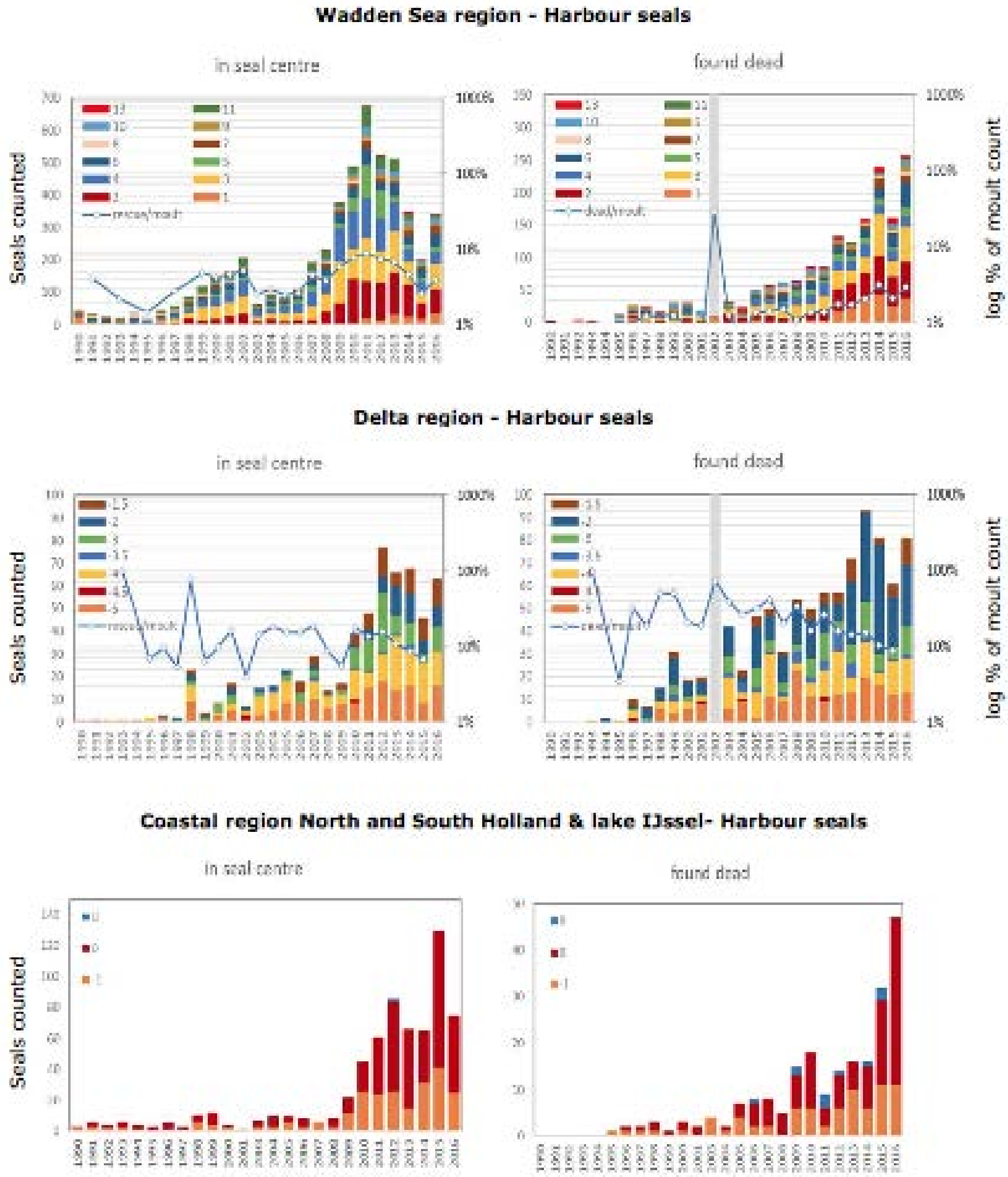

Figure 10. Annual numbers of harbour seals stranded alive and taken into seal centres (left) or found dead (right) in stacked bars for different areas. The total numbers stranded relative to the total moult counts are shown as a line (right axis; log scale) Top: counts in the Wadden Sea region of the Netherlands; Centre: counts in the Delta region Bottom: Coastal region of North and South Holland and the lake IJssel. From Brasseur (2018).

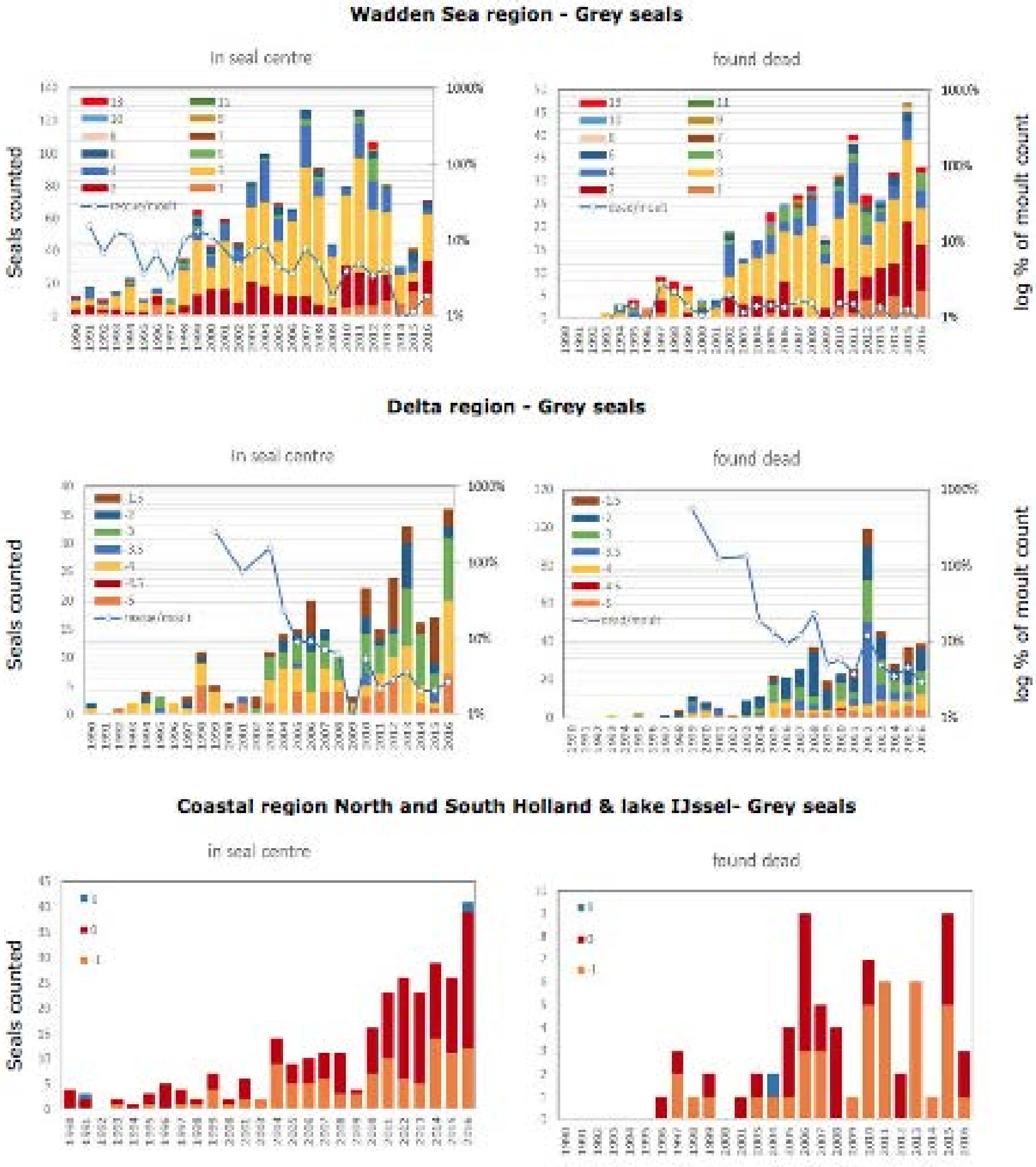

Figure 11. Annual numbers of grey seals stranded alive and taken into seal centres (left) or found dead (right) for different areas. Top: counts in the Wadden Sea region of the Netherlands; Centre: counts in the Delta region. Bottom: Coastal region of North and South Holland and the lake IJssel. From Brasseur (2018).

Between 1990 and 2016 the number of rescued animals was much higher than the number of seals found dead in the Netherlands. For example, in 2011 a total of 970 grey and harbour seals were taken into rehabilitation centres in the Netherlands, while 255 were found dead. As 90% of the rescued animals are pups, the analysis shows that the numbers of pups rescued often exceeded 50% of the total number of pups counted alive during surveys in the wild. In areas where immigration from other areas play an important role, i.e. for grey seals in the Wadden Sea and harbour seals in the Delta Area, numbers of pups rescued even regularly exceeded 100% of the numbers counted (see e.g. grey seals, Wadden Sea region, in 2004 and 2006).

Given the large numbers of stranded seals reported, the **WGMME support the recommendation** in Brasseur (2018) that it would be advisable to establish an official monitoring programme for seal stranding events, similar to the one that exists for cetaceans, instead of depending on data from the public. Although numbers of rescued animals have dropped significantly in the Wadden Sea in the last three years (2014–2016) to approximately 10% of the pup counts, this still represents a large group of animals; e.g. 628 animals were taken to rehabilitation centres in 2016, which cannot be defined as "the lowest level possible" agreed upon in the Trilateral Wadden Sea Agreement.

Brasseur (2018) noted that in the Netherlands rescued animals often are released after several months in the centres and the possible effect on the wild population have not been studied. The large numbers of released seals could affect the natural selection (Jensen *et al*., 2017), known and unknown diseases could be redistributed in the population (Stamper *et al*., 1998; Goldstein *et al*., 2004) and seals deprived of their youth in the wild could show unknown social defects affecting the population. Regardless of these issues, the current status of the populations in the Netherlands might give rise to new issues regarding seal rescue: As the populations approach carrying capacity, the rescued animals will cause an increase in demand of resources, resulting in unnaturally high exhaustion of available resources and, for example, accelerated rise in mortality. The rescue of pups could be a problem in a density-dependent population, as the natural mortality is expected to fluctuate responding to environmental drivers. Influencing pup survival might shift mortality to other animals (including adults), that would otherwise have survived.

2.1.2 Cetaceans

2.1.2.1 Passive acoustic monitoring (PAM) of harbour porpoises in the Baltic Sea

The Commission Decision (EU) 2017/848 lays out two primary criteria on the monitoring of abundance (D1C2) and distribution (D1C4) of small toothed whales. During the SAMBAH project (www.sambah.org), a methodology was developed for acoustic monitoring of harbour porpoises (*Phocoena phocoena*) in the Baltic Sea. Since the end of the project, several of the participating countries have initiated acoustic monitoring following the SAMBAH methodology, either as part of the national monitoring programme or on a project basis.

DENMARK: Denmark has initiated a monitoring program for the Baltic harbour porpoises in the waters around Bornholm. Ten CPOD stations, positioned on the original SAMBAH locations, will be deployed for one year, i.e. from June 2018 to June 2019. The plan is to repeat this monitoring at regular intervals, although the time frame for this has not yet been agreed.

FINLAND: Finland started acoustic monitoring of harbour porpoises in October 2016 and plans to continue at least until spring 2019. The monitoring is carried out in the northern Baltic Proper, in the offshore area south of Åland and the Archipelago Sea. C-PODs are deployed at 17 stations, eleven being former SAMBAH stations and six are additional stations between these. The C-PODs are serviced every 4–6 months. Preliminary results show the same seasonal pattern and similar detection rates as in SAMBAH. The monitoring is carried out by Turku University of Applied Sciences, funded by the Ministry of Environment and the Åland Government (Loisa, pers. Comm.).

GERMANY: The national acoustic monitoring started in 2002 (Gallus *et al*., 2012; Benke *et al*., 2014) and currently runs C-PODs at 15 stations in the German Baltic Sea (incl.

five at Fehmarn, two in the Kadet Trench, two northwest of Rügen, three at the Adlergrund and three on the Odra Bank in the Pomeranian Bay); four being former SAMBAH stations. Seven of these 15 stations are positioned within NATURA 2000 sites in the German EEZ. C-PODs are deployed 2–7 m below the water surface, which is different from deployment of C-PODs at SAMBAH stations, which was 2 m above the seabed. Stations are serviced every ten weeks and data are analysed continuously. During 2018 and 2019, additional stations will be equipped with C-PODS and further acoustic detectors (e.g. soundtraps) between Fehmarn and the Kadet Trench. The monitoring is carried out by the German Oceanographic Museum, funded by the Agency for Nature Conservation (Gallus, pers. Comm.).

POLAND: A pilot monitoring project supervised by Chief Inspectorate for Environmental Protection (CIEP, Poland) started in early spring 2016 and finishing data collection in March 2018. Monitoring is carried out by DHI Poland and Maritime Institute in Gdańsk. Two areas of C-POD deployment with five C-PODs each, previously included in the SAMBAH project, were chosen: Bay of Pomerania (including three CPODs in PLH990002/ PLB990003) and Stilo Bank (including three CPODs in PLB990002 and one CPOD in PLC990001). Preliminary results for the first season (2016–2017) show year-round occurrence of harbour porpoises in Pomerania Bay while Stilo Bank area is seldom visited by this species but the occurrence in this area show a clear seasonal pattern. All data collected within the pilot project will be analysed in 2018 and results will be publicly available at CIEP web page dedicated to habitats and marine species monitoring (http://morskiesiedliska.gios.gov.pl/pl/) and database (on request). Regular state monitoring scheme is planned for future (Malinga, pers. Comm.).

SWEDEN: A national acoustic monitoring programme of harbour porpoises started in spring 2017. The monitoring is commissioned by the Swedish Agency for Marine and Water Management (SwAM) and is carried out by the Swedish Museum of Natural History. C-PODs are deployed close to the bottom at ten former Swedish SAMBAH stations and one former Swedish BIAS station (www.bias-project.eu). Five of the stations are located within the Natura 2000 area 'Hoburgs bank' and 'Midsjöbankarna' (SE0330308) and six west thereof. The C-PODs are serviced around March–April and September–October, i.e. with an approximate service interval of six months. All C-PODs deployed in spring 2017 were retrieved in autumn 2017 and the data are currently being analysed. The data will be stored for public access at the Swedish Meteorological and Hydrological Institute (SMHI).

2.1.2.2 Visual monitoring and strandings

BELGIUM: In 2017, 93 harbour porpoises stranded, similar to the average of the last ten years (Figure 12). Important causes of death were predation by grey seals and incidental bycatch. Average densities recorded during aerial surveys ranged between 0.35 and 1.7 animals per km^2. The high rate of mother-calf pairs observed during a survey on 1–2 June was remarkable (Haelters *et al.*, 2018a).

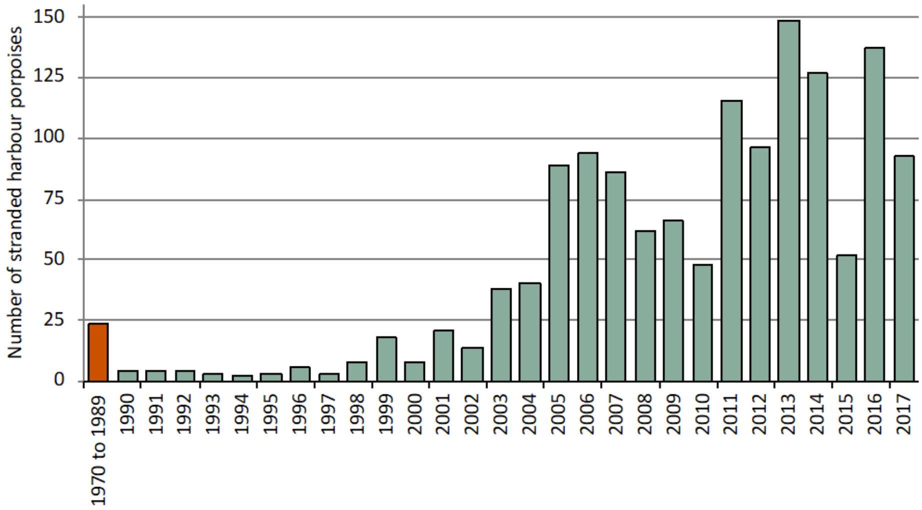

Figure 12. Strandings of harbour porpoises in Belgium recorded annually from 1990 to 2017 (plus total for 1970–1989). Data from Haelters *et al*. (2017; 2018a) and Haelters (unpubl.).

Strandings of a white-beaked dolphin (bycaught), and of a decomposed bottlenose dolphin were also recorded.

The most remarkable sighting in 2017 and, together with the dead narwhal of 2016 (Haelters *et al*., 2018b), the most remarkable ever in Belgium, was one of a bowhead whale close inshore in March–April (Figure 13). This might have been the first record of a bowhead whale ever in the North Sea (Haelters, 2017). The animal was entangled in fishing gear and/or rope. The animal was observed briefly again, shortly afterwards, off The Netherlands (Zeeland), and was not seen again afterwards.

Figure 13. Bowhead whale, *Balaena mysticetes*, sighted in the southeastern North Sea in 2017. Picture from Daan Drukker.

GERMANY: A survey of the German North and Baltic Sea was carried out between March and August 2016. Using aerial line transect surveys, a total of 114 harbour porpoise groups (129 animals) were recorded along 973 km of effort (during the spring months March to May near Borkum Reef Ground), and a total of 139 groups (175 animals) were recorded across the North Sea during summer (June–August) along 2456 km. In the southwestern Baltic Sea, in an area between the islands of Fehmarn and

Rügen, a total of ten porpoise groups (11 animals) were recorded along 1972 km of survey effort during summer. Due to logistic reasons, the western parts of the Baltic Sea and the eastern part of southern German Wadden Sea could not be covered (see Figure 14a&b).

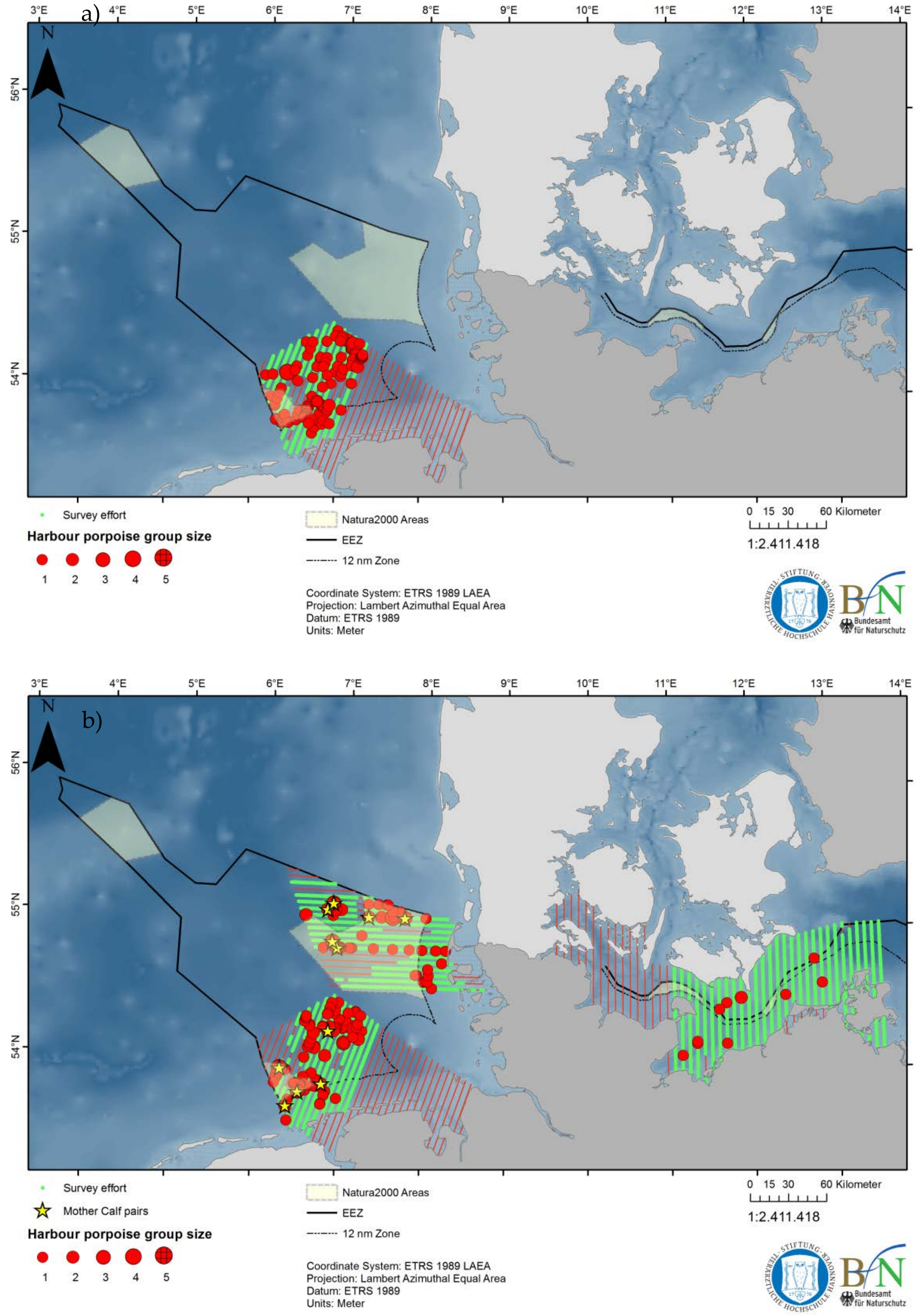

Figure 14. Survey effort and harbour porpoise sightings during aerial surveys in the German North Sea during a) spring 2016 and b) summer 2016. Harbour porpoise group sizes are indicated using group size dependent red circles; stars mark mother calf pairs; red lines indicate transect lines that were not covered though planned; green lines indicate covered transect lines.

Effort corrected density and abundance estimates were generated using a bootstrapping approach. The spring abundance for the Borkum area, southwest of the German Bight, was 6366 (95%CI: 3582–10 970) animals at 0.91 (0.51–1.56) animals / km². The same area yielded 6651 (3343–12 587) animals and 0.95 (0.48–1.79) animals / km² in summer. The area of Sylt Outer Reef, northeast of the German Bight, was estimated at 5779 (1535–13 439) animals and 0.72 (0.19–1.68) animals / km² during the summer of 2016. In the southwestern Baltic Sea, we estimated a total of 586 (147–1297) animals at an average density of 0.04 (0.01–0.10) animals / km² (excluding the area in the western Kiel Bight; Table 5).

Table 5. Summary of effort corrected, bootstrapped density and abundance estimates for spring and summer 2016 in the German EEZ of the North and Baltic Seas. N = estimated abundance of harbour porpoises; $N_{95\%CI}$ = 95% confidence interval around N; D = density estimate of harbour porpoises in ind./km²; $D_{95\%CI}$=95% CI around D; s = average group size.

area	season	N	$N_{95\%\ CI}$	D	$D_{95\%\ CI}$	$\hat{s}$
Borkum	spring	6366	3582–10 970	0.91	0.51–1.56	1.13
Sylt Outer Reef	summer	5779	1535–13 439	0.72	0.19–1.68	1.38
Borkum	summer	6651	3343–12 587	0.95	0.48–1.79	1.19
Fehmarn	summer	473	147–973	0.10	0.03–0.21	1.14
east of Fehmarn	summer	38	0–128	0.02	0.00–0.05	1.00
Pommeranian Bay	summer	75	0–196	0.01	0.00–0.04	1.00

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 2017 and will do so in 2018.

In 2017, a specific survey was dedicated to estimate marine mammal and seabird relative abundance and distribution in the area of Dunkerque (northern France) before construction of an offshore windfarm. The survey effort covered 9400 km² distributed as follows: 37% in France, 37% in Belgium and 26% in UK (Figure 15). Observations were conducted following a standardised protocol designed for aerial surveys (Laran *et al*., 2017). Four sessions were realised on 6–7 April (1526 km), 13–14 June (1534 km), 7–8 August (1532 km) and 4–5 December (1463 km). Two more sessions are planned in 2018 (Laran, per. comm.).

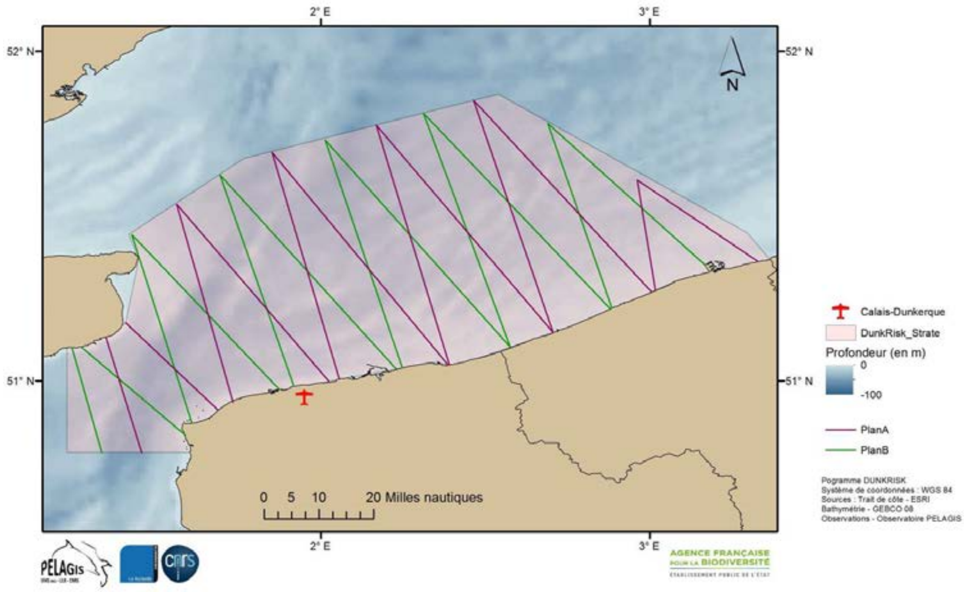

Figure 15. Total survey effort and transects in the study area (Pelagis data).

The total number of marine mammal observations was 323 during the first session, 115 during the second one, 45 during the third one and 221 during the fourth one (Table 6). Three species of cetacean have been observed including white-beaked dolphin (*Lagenorhyncus albirostris*) observed during session 3 and 4 and bottlenose dolphin (*Tursiops truncatus*) observed only twice during session 4. The harbour porpoise is the third species and the number of observations reflected a high seasonality for this species (Table 6). Harbour porpoise distribution also differed according to the sessions (Figure 16). During session 1, the species has been particularly present offshore Dunkerque, in the Belgium waters and at the frontier between the Belgium and French waters. The number of observations was more regularly distributed during the session 2. However, their numbers were higher offshore Dunkerque and in more offshore UK waters. Two other sessions are planned in 2018 and complete results will constitute the pre-construction assessment survey.

Table 6. Observations of marine mammals during the aerial survey (number of detections: N.obs, number of individuals: N.ind.), Pelagis data.

	Session 1 (06.-07.04.17)		Session 2 (13.-14.06.17)		Session 3 (07.-08.08.17)		Session 4 (04.-05.12.17)	
Groups/species	N.obs.	N.ind.	N.obs.	N.ind.	N.obs.	N.ind.	N.obs.	N.ind.
Cetacea	0	0	1	1	0	0	0	0
Lagenorhyncus albirostris	0	0	0	0	1	5	1	2
Phocoena phocoena	315	373	100	128	35	42	202	269
Tursiops truncatus	0	0	0	0	0	0	2	2
Phocidae	6	6	12	22	3	3	13	14
Halichoerus grypus	2	2	2	2	6	6	3	3
TOTAL	323	381	115	153	45	56	221	290

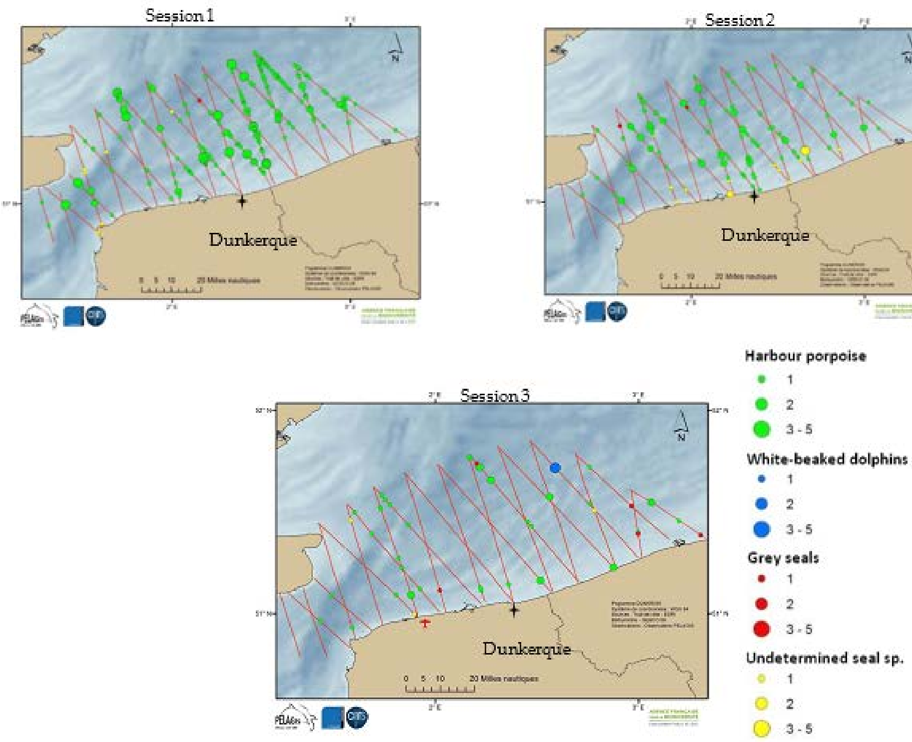

Figure 16. Observations of marine mammals during session 1 (6–7 April 2017), session 2 (13–14 June 2017) and session 3 (7–8 August 2017). Map for session 4 is not yet available. Pelagis data.

The integrated ecosystemic PELGAS ("Pélagiques Gascogne") survey carried out every year during spring in the Bay of Biscay has allowed studying changes in marine mammals' relative abundance. The study was carried out at the community level over more than a decade (2004–2016) (Authier *et al*., in press) and is situated within the larger MFSD subregion "Bay of Biscay and the Iberian coast". It is located between 43.5 and 48.5°N and includes mainly shelf waters (Figure 17).

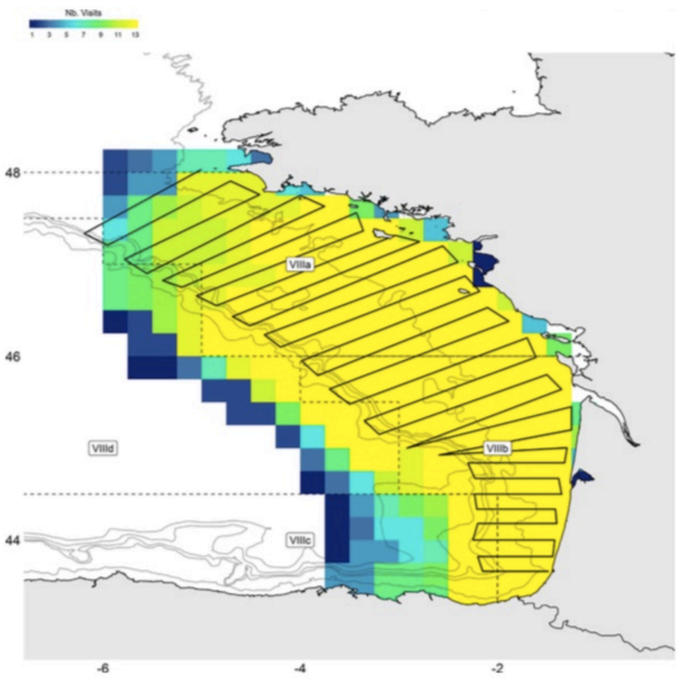

Figure 17. Map of the studied area in the Bay of Biscay. Colour shading codes how many year a given 0.25° × 0.25° block was visited between 2004 and 2016. Isobaths are depicting in light grey. ICES statistical areas in the Bay of Biscay are lineated by dashed grey lines. Black lines represent the survey transects (Authier *et al*., in press).

The relative abundance of the twenty-three most frequently sighted species (six cetaceans and 17 seabirds) was estimated by distance sampling and averaged over the study period and area. Cetacean species included five Delphinidae (common dolphin, bottlenose dolphin, striped dolphin, pilot whale and Risso's dolphin) and one Balaenopteridae (minke whale). Temporal changes were investigated with a Dynamic Factor Analysis (see Authier *et al*., in press; for more details on the exploratory statistical techniques used). Overall, cetacean species were more abundant in the southern part of the Bay of Biscay. The relative abundance of cetacean species slightly increased between 2004 and 2016 (Figure 18).

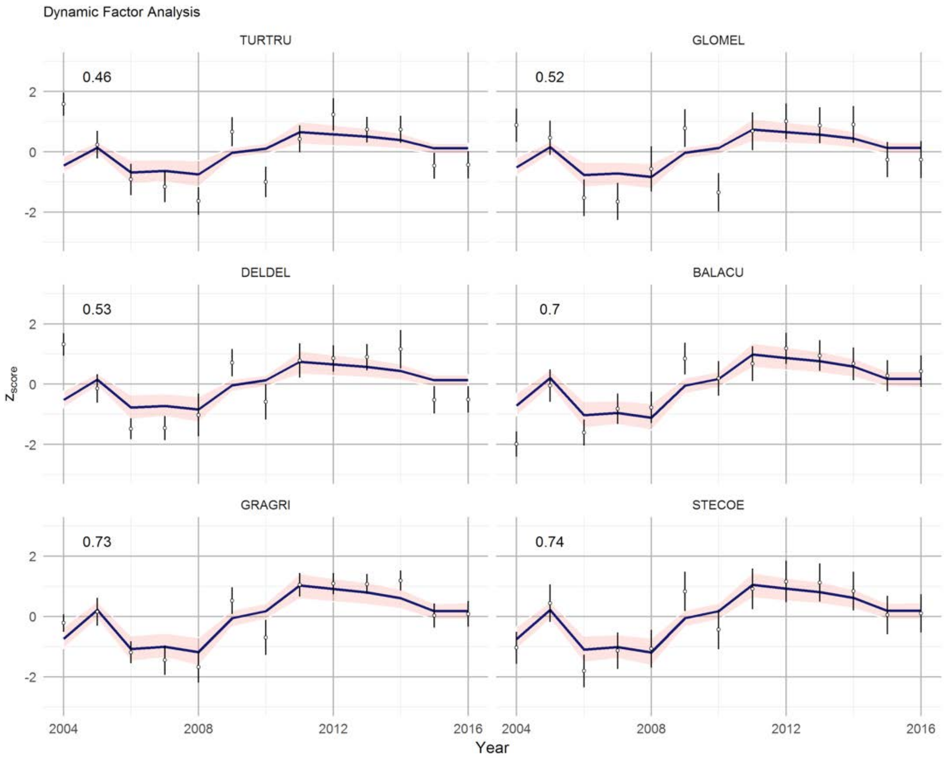

Figure 18. Dynamic Factor Analysis of z-scored time-series cetacean relative abundance in the Bay of Biscay. Species code are TURTRU for *Tursiops truncatus*, GLOMEL for *Globicephala melas*, DELDEL for *Delphinus delphis*, BALACU for *Balaenoptera acutorostrata*, GRAGRI for *Grampus griseus*, STECOE for *Stenella coeruleoalba*. See Authier *et al.*, in press, for further details.

FRANCE/Strandings: The French National Stranding Network (Réseau National d'Echouage, RNE) is the main tool for monitoring marine mammal stranding. Strandings have been recorded since the early 1970s in France. The network is considered to be relatively stable since the 1980s with consistent reporting since the early 1990s, in particular following the publication of a ministerial circular. Beyond 1990, it is therefore assumed that observed fluctuations or trends reflect biological or physical parameters such as abundance, mortality or drift conditions. The total number of cetacean strandings in 2016 was 1342. This is well above the average of the last ten years, estimated at 820 strandings per year (red line Figure 19). This historical series, despite some fluctuations, shows an overall tendency to an increase of strandings along French coasts. Despite a smaller number of strandings in 2015, the number of the year 2016 reached a new record (Dars *et al.*, 2017).

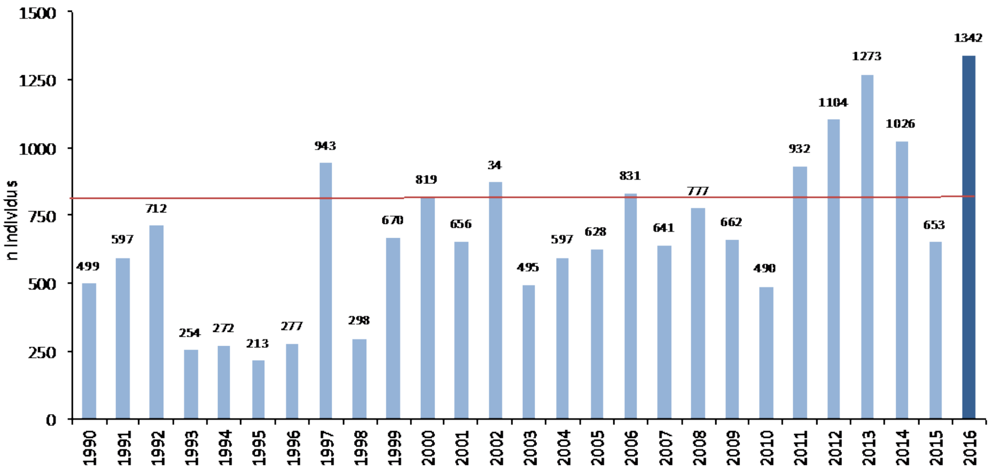

Figure 19. Annual distribution of the number of cetacean strandings along the French coasts from 1990 to 2016 (n = 18 533) (Dars *et al.*, 2017).

Thirteen species of cetaceans were observed in 2016 (Figure 20) and the relative abundances revealed a typical composition with the presence of the so-called regular species. Among the dominant species, the common dolphin is the most represented species, with 53.3% of strandings, followed by the harbour porpoise (31.3%). The percentage of individuals exhibiting bycatch marks was 42% for common dolphins (530 individuals examined), 37% for harbour porpoises (297 individuals examined), 21% for bottlenose dolphins (34 individuals examined) and 14% for striped dolphins (86 individuals examined) (Pelagis data).

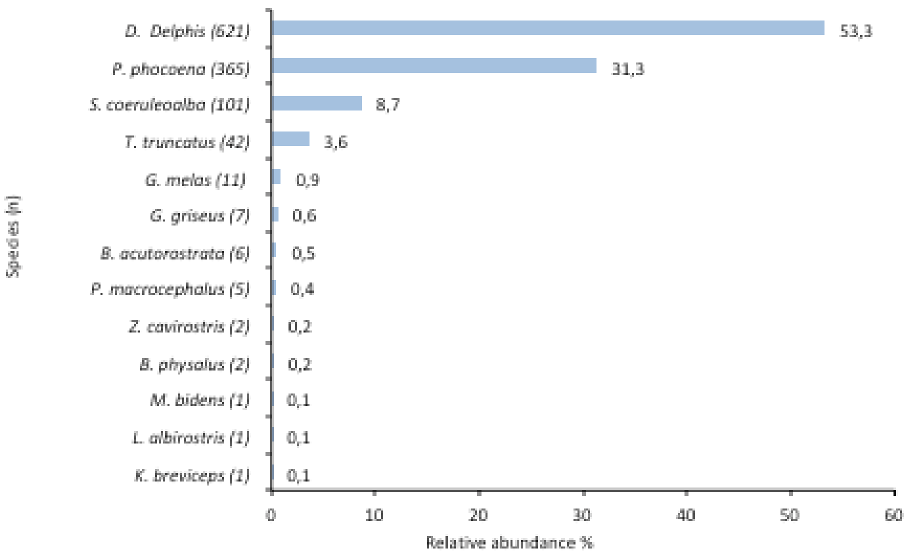

Figure 20. Relative abundance of the cetacean species stranded along the French coasts in 2016 (n = 1165; 177 undetermined individuals were excluded) (Dars *et al.*, 2017).

During 2017, two unusual multiple stranding events occurred in February–March along the French Atlantic coast. In total, ca. 700–800 common dolphins stranded, of which 80% exhibited bycatch marks. This is a pertinent reminder of the significance of the bycatch issue for common dolphin in the eastern North Atlantic. This has recently been reported to the IWC SC (Peltier *et al*., 2017; also see ToR C this report).

FRANCE: **Information on the importance of energy-rich foodwebs in the Bay of Biscay to sustain marine mammal populations**. Spitz *et al*. (in press) undertook a comprehensive assessment of energy requirements and prey consumption for the ten most abundant cetacean species (harbour porpoise, common dolphins, striped dolphins, bottlenose dolphins, long-finned pilot whales, Risso's dolphins, sperm whales, Cuvier's beaked whales, minke whales and fin whales) in the Bay of Biscay by combining recent data on their abundances from aerial surveys, and diets from stomach content analyses. Spitz *et al*. (in press) studied the trophic web through functional considerations to group prey and addressed interspecific differences in the cost of living of cetaceans that are independent of body size. Twelve functional prey groups that share similar key functional traits related to predatory characteristics of cetaceans were identified. The results show that small energy-rich schooling fish were the key prey group sustaining a large part of the cetacean community in the Bay of Biscay. The biomass removal of small energy-rich schooling fish by cetaceans is six times higher than removals of all other prey groups (between 272 900 and 465 300 tons/year). High quality nutritional resources appear to be crucial to sustaining cetaceans and maintaining ecosystem functions and services in the Bay of Biscay and the authors recommend that this should be carefully monitored.

IRELAND: **The ObSERVE Programme.** In 2014, a significant data acquisition programme was initiated with the aim of improving the information available on seasonal abundance, distribution and habitat use of cetaceans and seabirds needed to assess the potential impacts of human industrial activities (e.g. oil & gas, renewable developments and fisheries). Specifically, for cetaceans, a total of eight static and six towed acoustic surveys were undertaken in selected Atlantic Margin waters between 2015 and 2016, in addition to four combined line-transect and strip-transect aerial surveys. A total of 13 species were recorded acoustically including five mysticetes and eight odontocetes. A total of nearly 3.8 million detections of echolocation clicks and 375 000 tonal whistles were collected with AMARs and over 24 million candidate clicks detected by PAMGuard during PAM, resulting in a total of 1322 'cetacean events'. Blue whale infrasonic moans were only recorded in summer and autumn. Fin whale detections occurred at all moorings and in all seasons, with mean detection counts per hour lowest in summer and highest in autumn. Sperm whale clicks were detected at all moorings in all seasons but lowest at the two most southerly stations in all seasons. The number of sperm whale clicks per day varied significantly with month and season, with a northerly movement from spring to autumn and with more detections during night-time. Sowerby's and Cuvier's beaked whale clicks were recorded at all moorings in all seasons, with the highest rate for Sowerby's at the most northerly station in spring and for Cuvier's in spring at southern stations, which was the opposite of Sowerby's. Northern bottlenose, minke and sei whales were only occasionally detected.

During 2015/2016, 8700 km of aerial line transect surveys in Beaufort seastate ≤4 were conducted in both the summer and winter of 2015/2016. In summer, 12 cetacean species (ten odontocetes, two mysticetes) were identified and in winter 14 species (eleven odontocetes, including beluga, and three mysticetes). Abundance and density were estimated for harbour porpoise, common and bottlenose dolphin, and pilot, beaked and minke whales. There were clear seasonal differences in habitat use and abundance for common dolphin, which showed a fivefold increase in abundance in winter compared with summer. Similarly, for bottlenose dolphin, there was an eightfold increase in abundance during winter compared to summer. In contrast, fewer minke whales and harbour porpoise were recorded in winter than in summer. Fin whales were recorded during both seasons, suggesting a more complex distribution/movement pattern than simply migration. Deep-diving species, including pilot whales and at least four species of beaked whale (Ziphiidae) were recorded in relatively large numbers in both seasons over the continental slope and deeper waters, suggesting some habitat fidelity, especially in more northerly areas. This is the first time that abundance across a range of cetacean species has been estimated in this Atlantic region in winter, providing a baseline for future management and conservation efforts. It is expected that more detailed results of this programme will become available later in 2018.

IRELAND: **Beaked whales and other deep divers.** A variety of different studies in Ireland have suggested that the deep areas of the Rockall Trough, Porcupine Bight, and slope systems off the northwest of Ireland, where bathymetry is usually >1000 m depth provide important habitats for beaked whales (Boisseau *et al*., 2011; Wall *et al*., 2013; Oudejans, 2014; Rogan *et al*., 2017). Rogan and Hernandez-Milian (2011) reported 132 records of five Ziiphidae species from 1800 to 2009. Improving the knowledge of the distribution and abundance of this poorly known group of deep diving cetaceans is an essential prerequisite to inform mitigation strategies seeking to minimize their spatial and temporal overlap with noisy human activities which can result in potential population impacts (e.g. Balcomb III and Claridge, 2001; Jepson *et al*., 2003; Tyack *et al*., 2011; DeRuiter *et al*., 2013). Rogan *et al*. (2017) provides for the first abundance estimates for five deep diving cetacean species (sperm whale, long-finned pilot whale, northern bottlenose whale, Cuvier's beaked whale and Sowerby's beaked whale) using data from three dedicated cetacean sighting surveys that covered the oceanic and shelf waters of the Northeast Atlantic. Density surface modelling was used to obtain model-based estimates of abundance and to explore the physical and biological characteristics of the habitat used by these species. Distribution of all species was found to be significantly related to depth, distance from the 2000 m depth contour, the contour index (a measure of variability of the seabed) and sea surface temperature. Predicted distribution maps also suggest that there is little spatial overlap between these species. Rogan *et al*. (2017) constitute important baseline information to guide future risk assessments of human activities on these species, evaluate potential spatial and temporal trends and inform EU Directives and future conservation efforts.

NETHERLANDS: In July 2017, Wageningen Marine Research conducted aerial surveys to estimate the abundance of harbour porpoise on the Dutch continental shelf (Geelhoed *et al*., 2018). These surveys followed predetermined track lines in four areas: A - Dogger Bank, B - Offshore, C - Frisian Front and D - Delta. Between 7 and 18 July 2017 the entire Dutch continental shelf was surveyed, resulting in a total distance of 2362 km on effort. Of this covered effort, 1901 km (80.5%) was surveyed with good or moderate sighting conditions on at least one side of the plane. Marine mammals were assessed using line-transect distance sampling methods and density and abundance estimates were produced. In total, 230 sightings of 299 individual harbour porpoises were collected. Porpoise densities varied between 0.14–1.28 animals/km^2 in the areas A–D. The overall density on the entire Dutch continental shelf was 0.79 animals/km^2. (Table 7).

The total number of harbour porpoises on the Dutch continental shelf (areas A–D) was estimated at 46 902 animals (95% CI = 24 389–93 532) in July 2017 (Table 7). This number is in the same order of magnitude as the abundance estimate of 41 299 animals (95% CI = 21 194–79 256) in 2015 and lies in between the abundance estimates in July 2010 (N=25 998; 95% CI = 13 988–53 623) and July 2014 (N=76 773, 95% CI = 43 414–154 265).

Table 7. Density and abundance estimate of harbour porpoises in July 2017 per area on the Dutch continental shelf (DCS).

Area	Density (ind./km²)	95% CI	Abundance (n animals)	95% CI	CV
A – Dogger Bank	0.14	0.01–0.29	1325	167–2833	0.46
B – Offshore	1.28	0.55–2.92	21 584	9229–49 331	0.44
C – Frisian Front	0.53	0.08–1.53	6360	991–18 402	0.64
D – Delta	0.85	0.41–1.66	17 631	8595–34 552	0.37
Total DCS	0.79	0.41–1.86	46 902	24 389–93 532	0.35

Harbour porpoises were widely distributed and showed a homogenous distribution in a band from area D – Delta north to area B – offshore (Figure 21). The highest densities were found NW of the Wadden Isles. Harbour porpoises were virtually absent in large areas in the eastern part of area C – Frisian Front north of the Wadden Isles. Porpoises were scarce in area A – Dogger Bank.

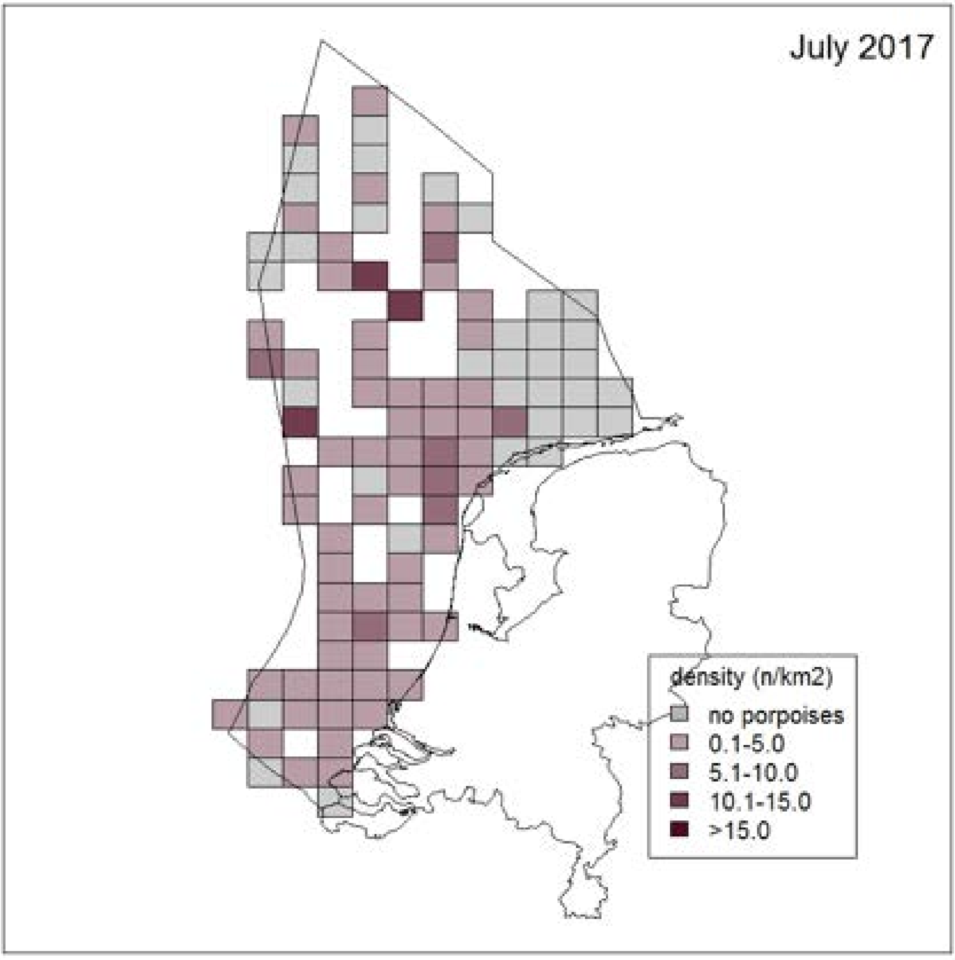

Figure 21. Density distribution of harbour porpoises (ind./km²) on the Dutch Continental shelf per 1/9 ICES grid cell, July 2017. Grid cells with low effort (<1 km²) were omitted.

In 2017, 695 stranded cetaceans divided over nine species were recorded by Naturalis (www.walvisstrandingen.nl). The harbour porpoise was the most abundant species (n = 683, Table 8). Since 2016, fresh harbour porpoises have been collected for post-mortem examinations by the Faculteit Diergeneeskunde, University of Utrecht. One of the main objectives of the research is to quantify human-induced causes of death. In 2017, 55 dead harbour porpoises were examined: 25 males and 30 females, 22 adults, 26 juveniles and seven neonates. Most of the examined harbour porpoises died as a result of infectious diseases (36%), bycatch (20%) or grey seal attacks (18%). The proportion of animals dying of infectious diseases was higher in 2017 than in previous years (IJsseldijk *et al*., 2018).

Table 8. Stranded cetaceans recorded in the Netherlands in 2017.

Species	N
Harbour porpoise	683
Minke whale	4
White-beaked dolphin	1
Striped dolphin	1
Common dolphin	1
Orca	1
Sowerby's beaked whale	2
Sperm whale	1
Fin whale	1
Total ind.	**695**

SWEDEN: Since 2016, the Swedish Museum of Natural History have collected up to 20 dead stranded or bycaught harbour porpoises per year for necropsies. The collection is funded by the Swedish Agency for Marine and Water Management and carried out in cooperation with Gothenburg Museum of Natural History, municipalities, other organisations and the general public. The necropsies are carried out in collaboration with the National Veterinary Institute. An overview of the harbour porpoises necropsied in 2016–2017 is given in Table 9. Samples are taken from all necropsied animals and stored in the environmental specimen bank at the Swedish Museum of Natural History. In addition, samples were taken directly in the field from another ten stranded harbour porpoises in 2016 and three in 2017 (Roos *et al*., 2017; 2018). Three of the necropsied harbour porpoises were first encountered as live strandings, but died during rescue attempts.

Table 9. Overview of harbour porpoises necropsied at the Swedish Museum of Natural History in 2016–2017. The numbers within brackets indicate animals that were first found as live strandings but died during rescue attempts.

Year	Sea				Cause of death				
	Skagerrak	Kattegat	Sound	SW Baltic	Bycatch	Other trauma	Disease	Starvation	Unknown
2016	5	2	3		4	1	3(1)		2
2017	2	8	8	2	9	1	3(1)	1	6(1)

UK: Since 2001, the Sea Watch Foundation has been monitoring the **bottlenose dolphin population inhabiting coastal waters of Cardigan Bay**, with annual summer abundance estimates, mainly using photo-ID capture–mark–recapture approaches, but also some line-transect Distance sampling (Lohrengel *et al.*, 2017). This monitoring effort has focused upon two Natura 2000 sites for the species, Cardigan Bay Special Area of Conservation in the south of the bay, and Pen Llyn a'r Sarnau in the north of the bay. Funding for the monitoring has come largely from Natural Resources Wales. The latest estimates (summer 2017) were 205 (95% CI 133–416) for the Cardigan Bay SAC and 258 (95% CI 161–508) for the wider Cardigan Bay using closed population models. The equivalent estimates using robust open population models were 157 (95% CI 119–252) for the Cardigan Bay SAC and 181 (95% CI 130–310) for the wider Cardigan Bay. Over the 17-year period, population size has fluctuated, but estimates in 2017 are not significantly different from those obtained in 2001.

UK: **Comparison of different survey methods for estimating harbour porpoise density.** Robust estimates of the density or abundance of cetaceans are required to support a wide range of ecological studies and inform management decisions. Considerable effort has been put into the development of line-transect sampling techniques to obtain estimates of absolute density from aerial- and boat-based visual surveys. Surveys of cetaceans using acoustic loggers or digital cameras provide alternative methods to estimate relative density that have the potential to reduce cost and provide a verifiable record of all detections. However, the ability of these methods to provide reliable estimates of relative density has yet to be established. Williamson *et al.* (2016) undertook such a comparison, with the primary aim of assessing whether measures of density obtained from PAM and digital surveys were reliable when compared with indices of density from conventional visual aerial surveys, for which robust correction to absolute density is possible. Secondary aim was to compare the performance of different acoustic metrics used to characterise variation in relative density and to provide a preliminary estimate of a scaling factor that can be considered as a proxy for the detection probability for aerial digital video surveys.

Estimates of relative density from visual surveys around acoustic monitoring sites were compared with several metrics previously used to characterise variation in acoustic detections of echolocation clicks in the Moray Firth. There was a strong correlation between estimates of relative density from visual surveys and digital video surveys. A correction to account for animals missed on the transect line, previously calculated for visual aerial surveys of harbour porpoise in the North Sea, was used to convert relative density from the visual surveys to absolute density. This allowed calculation of the first estimate of a proxy for detection probability in digital video surveys, suggesting that

61% (CV = 0.53) of harbour porpoises were detected. There was also a strong correlation between acoustic detections and density for detection positive hours.

UK: Diel variation in harbour porpoise habitat use. To ensure conservation and management measures are appropriately targeted, robust information on animal distribution and foraging behaviour is required. Often it is mainly visual survey data that are commonly used to model these distributions. Such data can only be collected in daylight and, therefore, modelled distributions and consequent management actions may not identify or protect important nocturnal habitats. Williamson *et al*. (2017) compared long-term passive acoustic data with visual survey data to reveal habitat-specific differences in diel patterns of detection in the Moray Firth. Harbour porpoises were detected consistently during night and day in sandy areas, with peaks in detection around sunrise and sunset, and at night in muddy areas (Figure 22). Detections also varied with depth, with the greatest proportion of daytime detections recorded in shallower sandy areas, and the most night-time detections recorded in deeper muddy areas. These findings suggest that the importance of muddy habitats could be underestimated when using visual survey data alone. This study highlights the value of using a combination of visual and acoustic methods where they are both available to characterise species distribution and to support efforts to develop spatio-temporal management of key habitats.

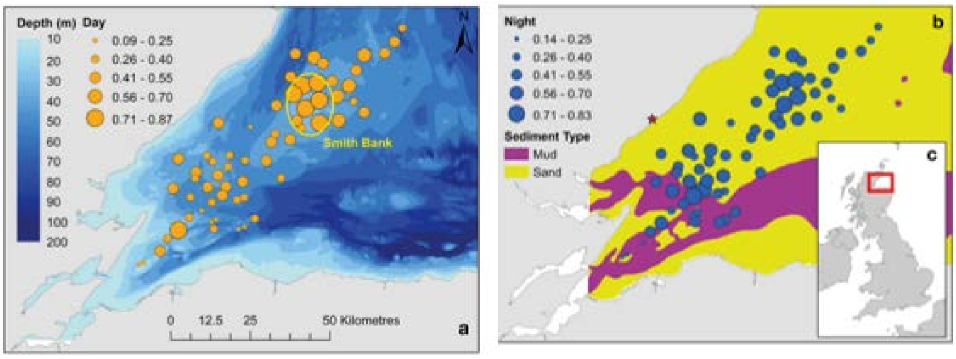

Figure 22. Locations of C-PODs showing the proportion of hours with detections of harbour porpoises during (a) day and (b) night. The background in panel (a) shows the bathymetry of the Moray Firth; Smith Bank is outlined in yellow. Panel (b) shows the two sediment types. Helmsdale (the location where sunrise and sunset data were obtained) is illustrated by the star in (b). (c) Location of the Moray Firth (red square) in relation to the British Isles.

UK: Categorizing click trains to increase taxonomic precision in echolocation click logger. Passive acoustic monitoring has provided invaluable insights into cetacean ecology. However, taxonomic classification of the echolocation clicks of odontocetes is an ongoing problem. Palmer *et al*. (2017) compared click train features logged by C-PODs to frequency spectra from adjacently deployed continuous recorders. A generalized additive model was used to categorize C-POD click trains into three groups: broadband click trains, produced by bottlenose dolphin (*Tursiops truncatus*) or common dolphin (*Delphinus delphis*), frequency-banded click trains, produced by Risso's (*Grampus griseus*) or white beaked dolphins (*Lagenorhynchus albirostris*), and unknown click trains. Using pooled model predictions, 98% of the click trains were classified and the predicted species distributions at 30 study sites matched well to visual sighting records from the region.

UK: Harbour porpoise movements in tidal streams. Evidence suggests that tidal stream conditions benefit top predators such as harbour porpoises presumably allowing them to optimise exploitation of prey resources (Johnston *et al*., 2005; Pierpoint, 2008; Bailey and Thompson, 2010). However, clear demonstration of this relationship is complicated by the fact that strong tidal flows often occur near-simultaneously across a wide area. The Great Race of the Gulf of Corryvreckan (western Scotland, UK) is a jetting tidal system where high-energy conditions persist across a broad range of tidal phases in a localised, moving patch of water (Figure 23). Porpoises can therefore actively enter or avoid this habitat, facilitating study of their usage of adjacent high- and low-energy environments. Benjamins *et al*. (2016) examined the distribution of harbour porpoises using passive acoustic porpoise detectors (C-PODs) deployed on static moorings (~35 d) and on Lagrangian drifters moving freely with the current (up to ~48 h). C-PODs moored in the path of the Great Race registered a significant increase in detections during the passing of the energetic tidal jet. Encounter durations recorded by drifting C-PODs were longer than those recorded by moored C-PODs, suggesting that porpoises tended to move downstream with the flow rather than remaining stationary relative to the seabed or moving upstream. The energetic, turbulent conditions of the Great Race are clearly attractive to porpoises, and they track its movement with time.

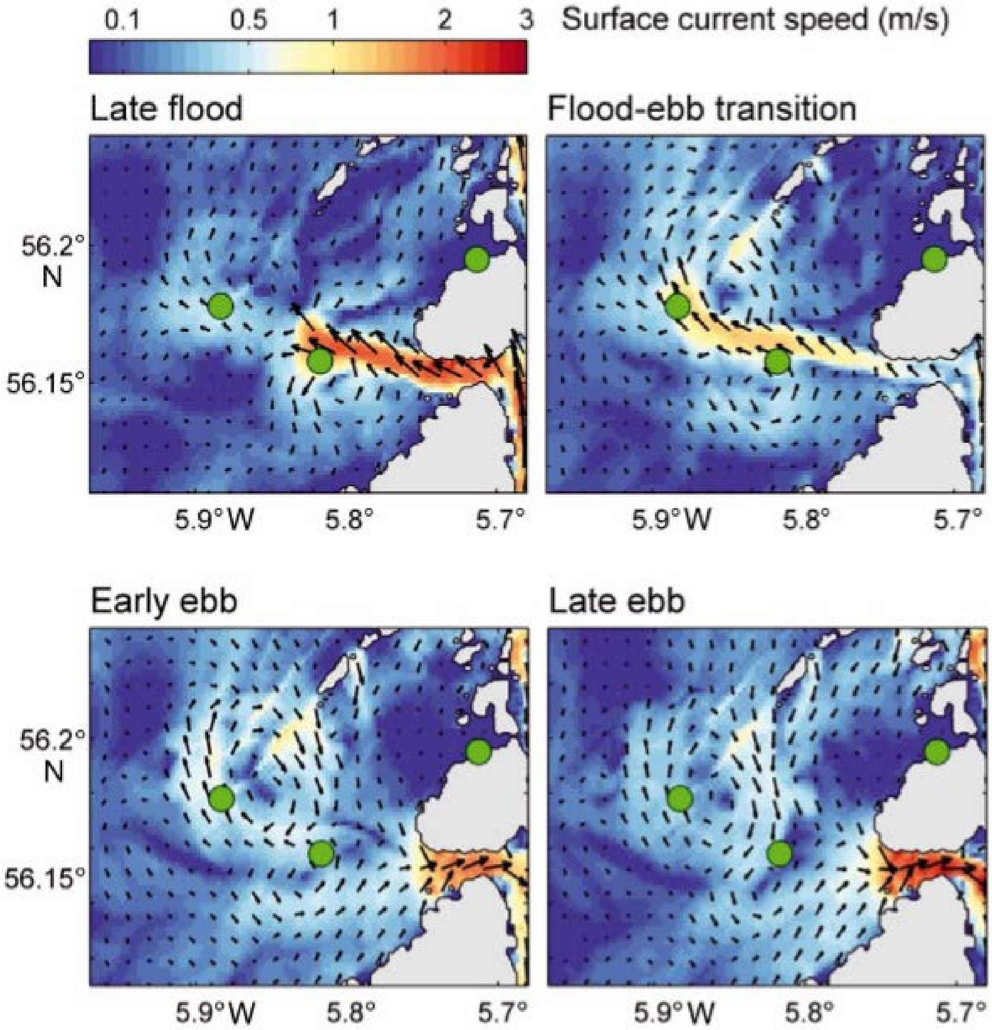

Figure 23. Surface currents from a hydrodynamic model of the Great Race relative to flood and ebb in the Gulf of Corryvreckan. The interval between panels is 2 h, and they correspond to approximately 67, 125, 183 and 241 tide-degrees relative to low tide in Oban. Vectors show current direction, and underlying colours show speed. Green discs correspond to mooring locations. (Taken from Benjamins *et al.*, 2016).

More recently, Benjamins *et al.* (2017) investigated small-scale variability of vocalising harbour porpoise distribution within two Scottish marine renewable energy development (MRED) sites using dense arrays of C-POD passive acoustic detectors (Figure 24). Daily detection rates varied significantly within both sites, with the modelling indicating linkages between porpoise presence and small-scale heterogeneity among different environmental covariates (e.g. tidal phase, time of day). Porpoise detection rates varied considerably but with coherent patterns between moorings only several hundred metres apart and within hours. These patterns are presumed to have ecological relevance. Benjamins *et al.* (2017) concluded that in energetically active and heterogeneous areas, porpoises display significant spatio-temporal variability of site use at scales of hundreds of metres and hours. Such variability will not be identified when using solitary moored PAM detectors (a common practice for site-based cetacean monitoring) but may be highly relevant to site-based impact assessments of MRED and other coastal developments. PAM arrays encompassing several detectors spread across a site appear to be a more appropriate tool to study site-specific cetacean use of spatio-temporally heterogeneous habitat and assess the potential impacts of coastal and nearshore developments at small scales.

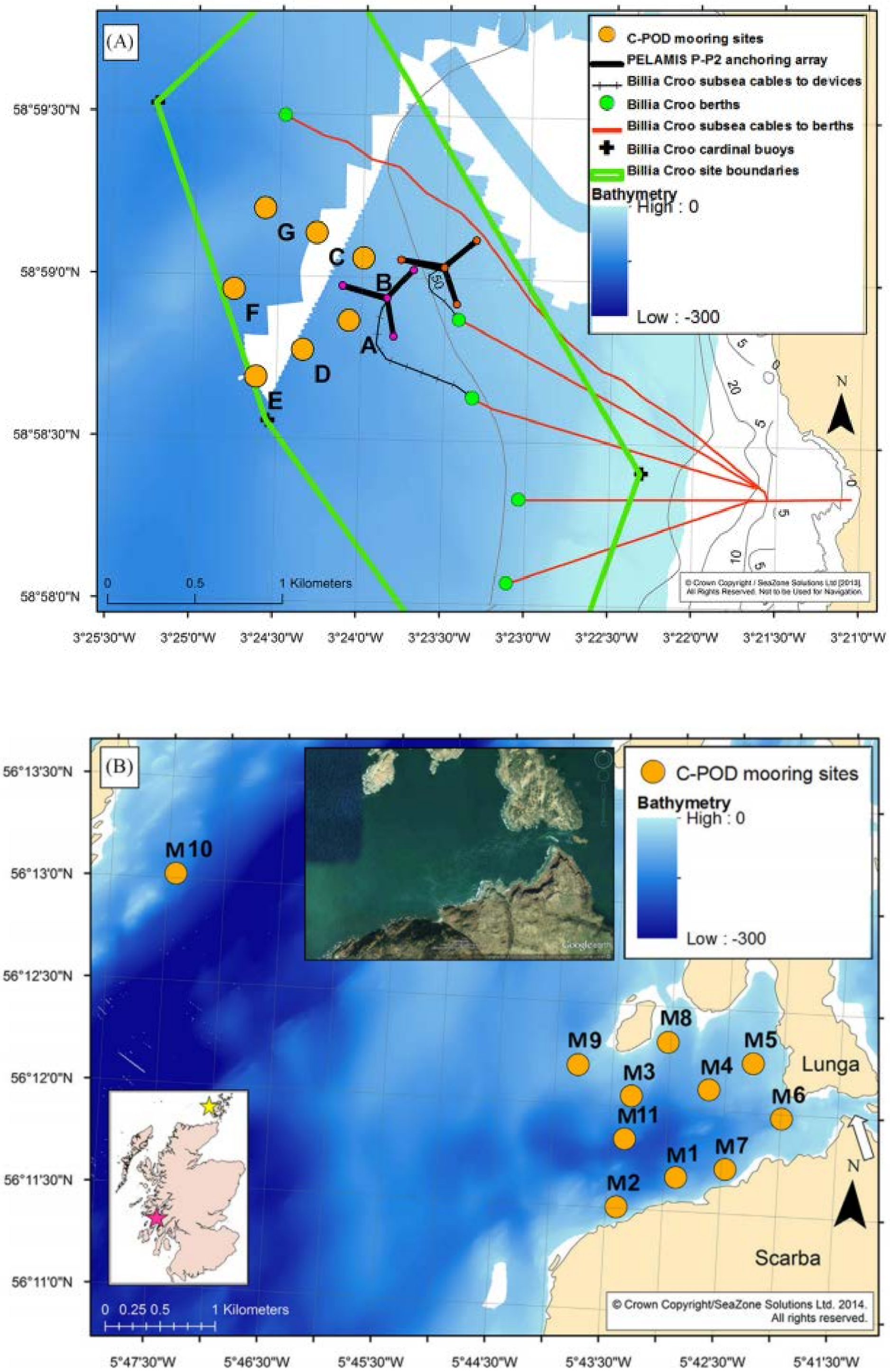

Figure 24. Overview of C-POD arrays. (A) Billia Croo (yellow star in inset map). EMEC test site boundaries, P-P2 anchoring assemblage and subsea cable infrastructure are also indicated (courtesy of EMEC). (B) Scarba (red star in inset map). Note Grey Dogs tidal channel (white arrow) to the east of Mooring 6, and example of westward tidal flow during flood tide (Google Earth inset). Bathymetry data were derived from the UK Hydro-graphic Office (Billia Croo) and from the INTERREG INISHydro project (Scarba). Gaps in bathymetry are shaded white. (Taken from Benjamins *et al.*, 2017).

NORTHWEST EUROPEAN SEAS: As part of a five-year Marine Ecosystems Research Programme, funded by the UK Natural Environment Research Council and Department of Food, Environment and Rural Affairs, Sea Watch Foundation/Bangor University have collated 2.5 million kilometres (137 000 hours) of cetacean survey effort from around fifty research groups in Northwest European seas over the period 1978–2016. Collectively, these surveys are being used to test ecological questions/hypotheses using a variety of modelling approaches, and to generate potentially useful data products. Using hurdle models that incorporate a range of environmental parameters believed to influence prey distributions and their availability for capture, maps of absolute densities of the twelve most common species are being produced at monthly temporal and 10 km spatial resolution across the past three decades (see general example in Figure 25 and a regional example in Figure 26 and Figure 27).

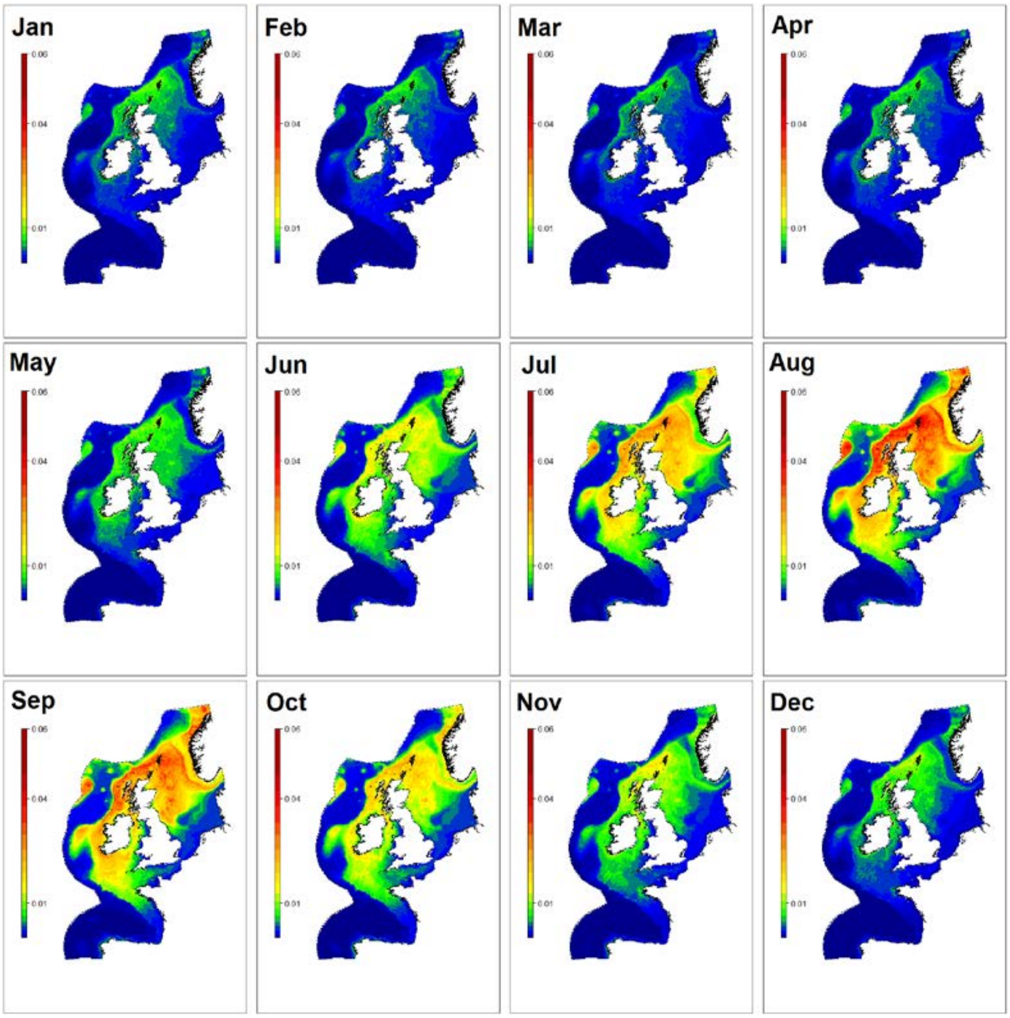

Figure 25. Predicted average monthly minke whale densities.

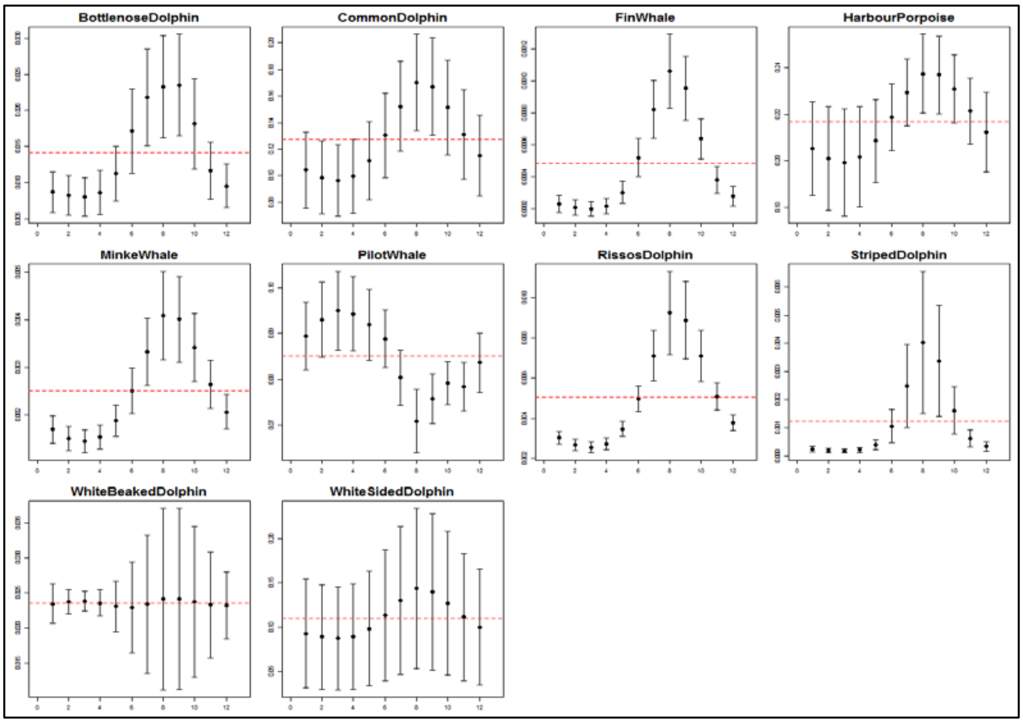

Figure 26. Sample regional seasonal trends for selected species: Scottish Hebrides.

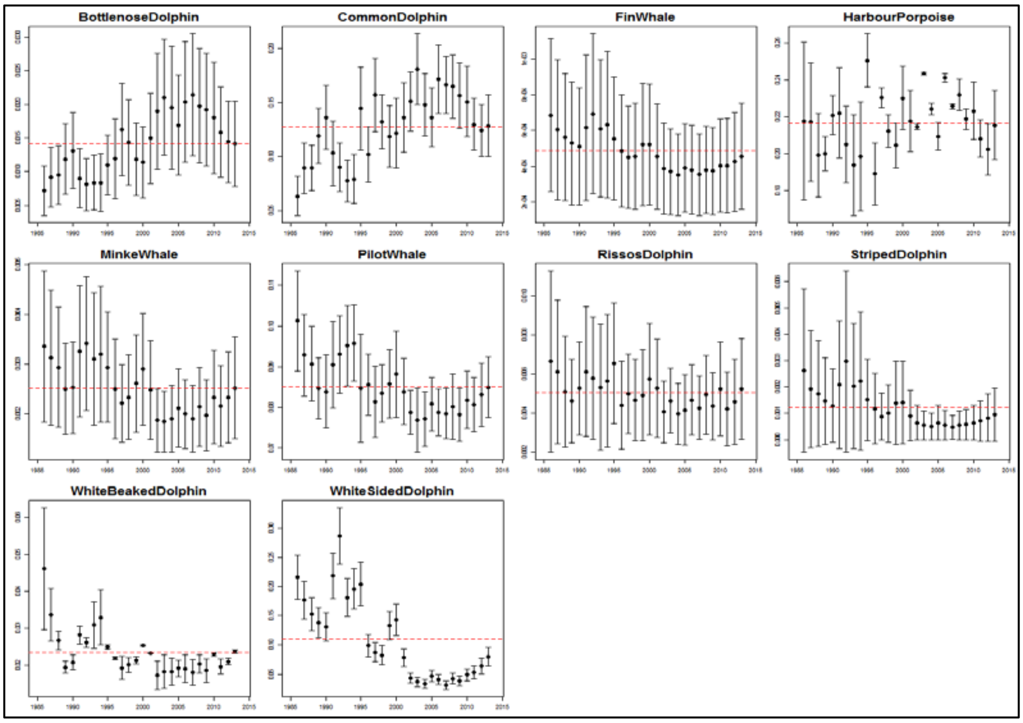

Figure 27. Sample regional interannual trends for selected cetacean species: Scottish Hebrides.

The outputs are being used to identify high density areas, at species and community levels, and to provide inputs for wider ecosystems models. In the final phase of the programme (2017–2018), risk mapping is being undertaken where monthly distributions are compared with those of different human activities, incorporating measures of the different vulnerabilities that a species faces from a particular activity.

North Atlantic: NAMMCO - Cetacean abundance and distribution in the North Atlantic. In 2017, at the Biennial Conference on the Biology of Marine Mammals, NAMMCO held a workshop on cetacean abundance and distribution in the North Atlantic with the aim:

- To generate a set of North Atlantic wide design-based abundance estimates for 2015/2016 for those cetacean species for which sufficient data are available. Species will include minke, fin, humpback, pilot whales and others that the data support. Estimates will be corrected for biases to the extent possible. The expected outcome is a complete set of estimates, or, more likely, an incomplete set of estimates and an action plan to achieve a complete set in timely fashion.
- To discuss modelling the spatial and temporal distribution and habitat use of cetaceans in the North Atlantic using data from 2015/2016. Discussion will be focused on the most important and available variables to inform modelling; the merits or otherwise of modelling the entire northern North Atlantic; the challenges of combining multiple datasets from different projects/platforms/methodologies; and the logistics and timelines of moving forward with modelling. The expected outcome is an action plan for moving forward (NAMMCO, 2018).

A North Atlantic-wide modelling of distribution and habitat use by cetaceans is considered valuable because:

1) It could help in understanding the large-scale distribution of several species, and why those distributions change over time.
2) Help predict future distribution based on predicted changes in the ocean environment.
3) Habitat modelling may identify areas that are likely to have large numbers of animals but which have not been adequately sampled to date.
4) Model-based abundance estimates are useful for comparison to design-based estimates and may be more precise and applicable to a smaller scale in some cases.
5) The modelling will help identify areas and times that are most susceptible to human impact; in some cases anthropogenic effects, for example noise production, could be included in some models.

It was also recognised that while the remit of the workshop was to look at 2015–2016 survey data, inclusion of older data could have benefits. For example, a larger dataset with better spatial coverage and more sightings would be advantageous and changes in distribution for several species are clearly apparent in the NASS and SCANS data and the environmental factors contributing to these changes are of interest and may improve the predictive ability of models. Priority species for modelling include the baleen whales: fin, blue, humpback and common minke. Harbour porpoise are abundant in shelf waters throughout the survey area and are a species of high interest particularly in relation to the impact of fisheries bycatch. Several species of dolphin, including white beaked, white sided, common and striped dolphins will also be included. During the workshop, the survey coverage and availability of data for the generation of a 2015/2016 abundance was documented (Figure 28).

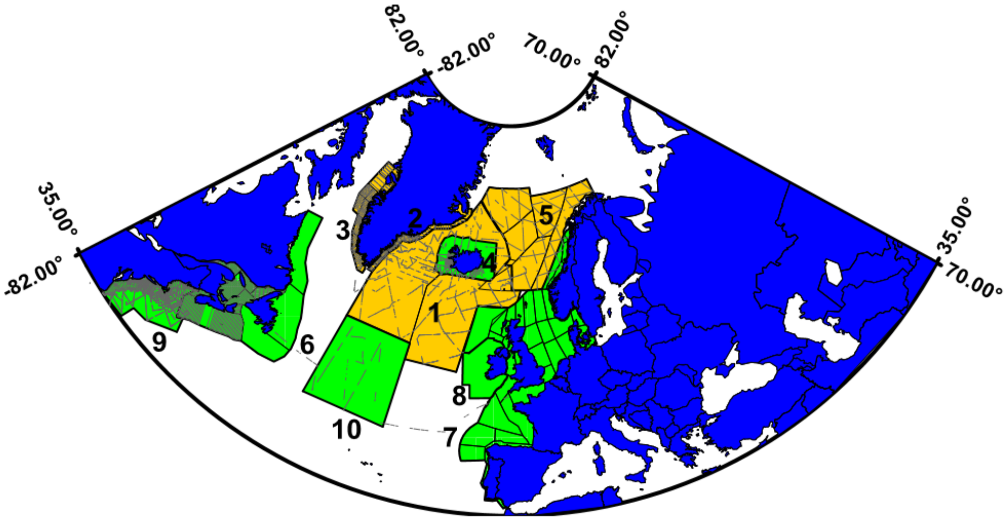

Figure 28. Survey coverage, 2015 (gold) and 2016 (green). Numbers refer to survey areas (taken from NAMMCO, 2018).

Considering the scale of this analysis, it was noted that while cetaceans of the same species are usually assumed to respond to their environment in the same way, there can be regional and/or population differences that may limit the usefulness of large-scale models. For example, Mannocci *et al.* (2017) noted that for some species, separate models for the eastern and western North Atlantic performed better than a combined model, suggesting that the cetaceans in the two areas, even of the same species, did not respond in the same way to their environment. It was agreed that the North Atlantic modelling work would begin at the largest scale to discern simple ecological correlates for each species, and then focus on the regional level with more complex and detailed models. It is likely that this project will take 2–3 years to complete.

2.2 New information on population/stock structure

Harbour porpoise (*Phocoena phocoena*)

Lah *et al.* (2016) investigated the population differentiation of harbour porpoises in European Seas with a focus on the complex population structure of porpoises from the Baltic Sea and adjacent waters, using a population genomics approach. Three main groupings at the level of all studied regions were suggested, (1) the Black Sea, (2) the North Atlantic and (3) the Baltic Sea. A distinct separation of the North Sea harbour porpoises from the Baltic Sea populations was observed and splits between porpoise populations within the Baltic Sea were identified. A notable distinction between the Belt Sea and the Inner Baltic Sea (IBS) subregions was detected. This genomic evidence is especially important since it underlines the importance that IBS population should be managed separately from the neighbouring Belt Sea population. This study supports evidence from Sveegaard *et al.* (2015), who defined management borders for harbour porpoises in the Baltic Sea by combining data from genetic, morphometric, acoustic and satellite tracking methods. The border between the North and Belt Sea populations was primarily based on the May–September distribution patterns of harbour porpoises equipped with satellite tags at the tip of Jutland and in inner Danish waters. The best fit was found for a latitudinal border at 56.95°N. The eastern management border of the Belt Sea population in the southern Baltic Sea was primarily based on the year-round longitudinal distribution pattern of harbour porpoises equipped with satellite tags in the inner Danish waters only, and harbour porpoise echolocation activity at a subset of the SAMBAH C-POD stations (between 12 and 15°E). This showed that from May to September 90% of their daily locations were west of 13.5°E. Similarly, there was a clear drop in acoustic detections east of this longitude. Thereby, the best management area for the Belt Sea population was proposed to extend from latitude 56.95°N in the northern Kattegat Sea to west of longitude 13.5°E in the south Baltic Sea.

Mikkelsen *et al*. (2016) applied and compared two different methods of describing the distribution pattern of harbour porpoises in the southern Baltic Sea. The density of daily positions derived from 13 harbour porpoises equipped with satellite transmitters in the inner Danish waters was modelled and compared to harbour porpoise echolocation activity recorded at 36 SAMBAH C-POD stations (all west of Bornholm). Both methods showed a significant linear relationship with a strong decline in porpoise occurrence from west to east, similar to the findings by Sveegaard *et al*. (2015).

In the SAMBAH project, a division between the Belt Sea and Baltic Proper populations was identified for management reasons. The division was based on the pattern of detections or no detections per station and month in the data, together with published information on satellite tagged harbour porpoises from the Belt Sea population (Sveegaard *et al*., 2015; Mikkelsen *et al*., 2016) and seasonal patterns in harbour porpoise echolocation activity in German Baltic waters (Benke *et al*., 2014). Published data on harbour porpoises annual reproductive cycle (Börjesson and Read, 2003; Lockyer and Kinze, 2003) were also taken into account to define two seasons consisting of six months each. This resulted in a proposed border following a diagonal line, approximately between Hanö in Sweden to Słupsk in Poland, during May–October (Carlén *et al*. in review).

The WGMME concludes that these above-mentioned studies confirm the existence of three harbour porpoise populations: The North Sea, the Belt Sea and the Baltic Proper (or Inner Baltic) population. Further, WGMME notes the difference between the summer borders for the Belt Sea and Baltic Proper porpoise populations in the southern Baltic Sea proposed by Sveegaard *et al*. (2015) and Carlén *et al* (in review), respectively. Nevertheless, the results are not necessarily inconsistent as the distribution pattern of the satellite tagged harbour porpoises does not say anything about the distribution pattern of Baltic Proper animals, and it is not possible to separate the populations acoustically. Rather, those results, together with the critical status of the Baltic Proper population and the lack of a spatial separation between the Belt Sea and Baltic Proper populations during winter, highlights the need for precautionary management of harbour porpoises in the Baltic Sea, and calls for further investigations of the population distribution ranges in the region.

Kesselring *et al*. (2017) investigated the onset of sexual maturity in female harbour porpoises in the period 1990–2016. Ovaries from 111 female harbour porpoises from the German North Sea (n=69) and Baltic Sea (n=42) were examined for the presence and morphological structure of follicles, *corpora lutea* and *corpora albicantia*. Kesselring *et al*. (2017) performed the first model-based estimation of age at sexual maturity for harbour porpoises from German waters and produced a demographical age structure based on all female strandings and bycatches. Using a model approach, the threshold was identified at which more than 50% of all specimens qualify as mature without setting an arbitrary threshold that is biased by the observer. The age of specimen ranged from 0 to 22 years, with a mean age of sexual maturity of 4.95 years. No significant differences between specimens from the North Sea and Baltic Sea were detected. However, the average age at death differed significantly with 5.70 (± 0.27) years for North Sea animals and 3.67 (± 0.30) years for those in the Baltic Sea. The female part of the North Sea population contains 45.34% above the threshold age of sexual maturity, while the Baltic Sea population contains only 27.56 % mature females. Growing evidence exists that the shortened lifespan of Baltic Sea harbour porpoises is linked to an anthropogenically influenced environment with rising bycatch mortalities due to local gillnet fisheries.

Fontaine *et al.* (2017) analysed the fine-scale genetic and morphological variation in harbour porpoises around the UK. Porpoises from the southwestern UK are genetically differentiated and have significantly larger body sizes compared to those of other UK areas. Southwestern UK porpoises showed admixed ancestry between southern and northern ecotypes with a contact zone extending from the northern Bay of Biscay to the Celtic Sea and Channel.

Bottlenose dolphin (*Tursiops truncatus*)

Female reproductive success and calf survival in a North Sea coastal bottlenose dolphin population. Between-female variation in reproductive output provides a strong measure of individual fitness and a quantifiable measure of the health of a population which may be highly informative to management. Robinson *et al.* (2017) used longitudinal sightings data over 20 years to examined reproductive traits in female bottlenose dolphins from the east coast of Scotland (Figure 29). From a total of 102 females identified between 1997 and 2016, 74 mothers produced a collective total of 193 calves. Females gave birth from six to 13 years of age with a mean age of eight. Approximately 83% (n = 116) of the calves of established fate were successfully raised to year 2–3. This calf survival rate is similar to that observed in other long-term bottlenose dolphin studies (e.g. 81% in Sarasota Bay, USA [Wells and Scott, 1990]; 86% in Doubtful Sound, New Zealand (Currey *et al.*, 2008). Of all known mortalities, more than 45% were attributed to primiparous females. Calf survival rates were also lower in multiparous females who had previously lost calves. A mean interbirth interval (IBI) of 3.80 years (n = 110) was recorded which is comparable to that recorded in other *Tursiops* populations (e.g. Wells *et al.*, 1987; Mann *et al.*, 2000; Kogi *et al.*, 2004; Steiner and Bossley, 2008; Henderson *et al.*, 2014) although Fruet *et al.* (2015) reported shorter IBIs of 3.3 years for animals from the Southwest Atlantic. Calf loss resulted in shortened IBIs, while longer IBIs were observed in females assumed to be approaching reproductive senescence.

Maternal age and size, breeding experience, dominance, individual associations, group size and other social factors, were all concluded to influence reproductive success in this population. As a result, some females are likely more important than others for the future viability of the population (Whitehead and Mann, 2000; Brough *et al.*, 2016). Consequently, a better knowledge of the demographic groups containing those females showing higher reproductive success would be highly desirable for conservation efforts aimed at their protection.

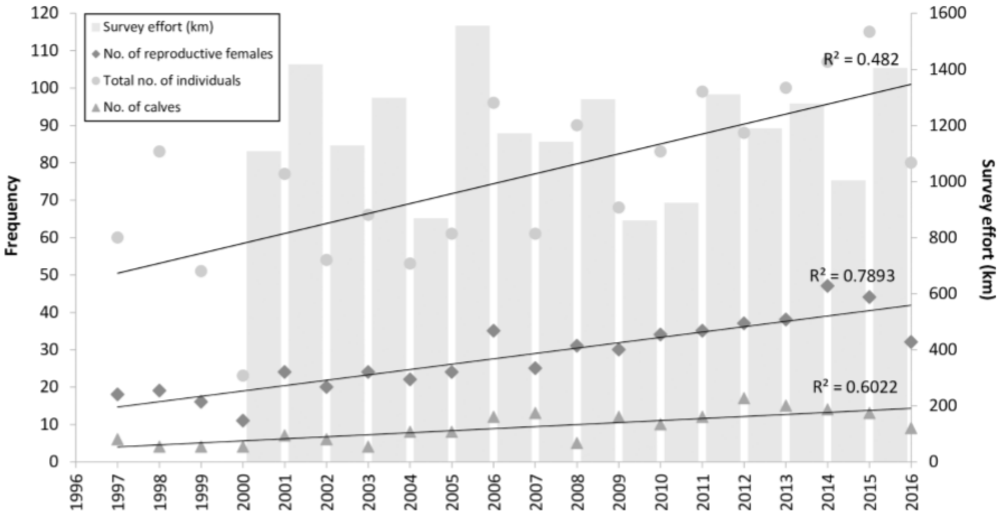

Figure 29. The total number of individuals, number of reproductive female bottlenose dolphins and the number of calves recorded in the outer southern Moray Firth study area from 1997 to 2016 inclusive. (Taken from Robinson *et al.*, 2017).

Bottlenose dolphin social structure is distinguished by age- and area-related associations. Social structure can affect population growth, genetics, and animal movements, and represents an important factor in management and conservation (Whitehead, 2008; 2009). Social relationships exhibited by individual dolphins are maintained within a constantly changing social environment where individuals are drawn from a large social network (where they may be present in a wide range of different groups) but associate consistently with just a few other individuals (Smolker *et al.*, 1992). Baker *et al.* (2017) analysed the social structure of bottlenose dolphins in the Shannon Estuary, Ireland. One hundred and twenty-one dolphins were identified during 522 sightings between 2012 and 2015. Baker *et al.*, (2017) reported that the bottlenose dolphins in the Shannon Estuary fit the general pattern of a fission-fusion society structured by age and area class, but perhaps one also characterized by unusual female–male associations. Although individuals in the population associate with many others in a complex social network, significantly strong, persistent and preferred associations exist between individual dolphins. Unlike other studies, there is little evidence in the Shannon Estuary population for adult male groups, female–calf groups or exclusively juvenile groups.

Movement analyses demonstrated the use of the inner Shannon Estuary by only 25% of the population revealing a potential community division by area. The use of the inner estuary by only a small percentage of the population seasonally has strong implications for management of the population as a whole. For example, the degree of exposure to anthropogenic threats would be different for individuals that largely resided in the inner estuary compared to those in the outer estuary. These results highlight the importance of localized research, reflecting the complexity found in bottlenose dolphin societies globally.

Sperm whale (*Physeter macrocephalus*)

Autenrieth *et al.* (2018) investigated the maternal relatedness and the putative origin of the 30 male sperm whales stranded in five different countries bordering the south North Sea in 2016. It has been postulated that these individuals were on a migration route from the north to warmer temperate and tropical waters where females live in social groups. Samples from four countries (n = 27) were included in this study that utilized existing genetic resources to sequence 422 bp of the mitochondrial control region, a molecular marker for which sperm whale data are readily available from the entire distribution range. Based on four single nucleotide polymorphisms (SNPs) within the mitochondrial control region, five matrilines could be distinguished within the stranded specimens, four of which matched published haplotypes previously described in the Atlantic. Among these male sperm whales, multiple matrilineal lineages co-occur. Results showed that the genetic diversity of these male sperm whales is comparable to the genetic diversity in sperm whales from the entire Atlantic Ocean and that within this stranding event, males did not comprise maternally related individuals and apparently included assemblages of individuals from different geographic regions.

Spain: Factors affecting stranding patterns. Strandings can provide valuable information on the presence and relative abundance of cetaceans in an area (e.g. López *et al.*, 2002; Siebert *et al.*, 2006; Leeney *et al.*, 2008; Truchon *et al.*, 2013), as well as allowing collection of samples to characterize life history, contaminant burdens, diet and feeding ecology, population genetics and other information for individuals and populations (López, 2003; Pierce *et al.*, 2008; Fontaine *et al.*, 2010; Fernández *et al.*, 2011; Murphy *et al.*, 2013; Méndez-Fernandez *et al.*, 2014; Santos *et al.*, 2014). Nevertheless, the relationship between stranding patterns and population dynamics is often unclear. Absolute abundance of cetaceans cannot be estimated directly from strandings because the mortality rate, the proportion of dead dolphins that reach the coast and the proportion of strandings found by strandings networks are all unknown. Saavedra *et al.* (2017) investigated the spatio-temporal patterns and trends in the numbers of strandings Galicia (NW Spain) and their relationships with meteorological, oceanographic, prey abundance and fishing-related variables. The strandings of 1166 common dolphins (*Delphinus delphis*), 118 bottlenose dolphins (*Tursiops truncatus*) and 90 harbour porpoises between 2000 and 2013 were incorporated into the analysis. Generalised additive and generalised additive-mixed model results indicated that the local ocean meteorology (strength and direction of the North– South component of the winds and the number of days with southwestern winds) and the winter North Atlantic Oscillation Index best explained the stranding pattern. There were no significant relationships with indices of fishing effort or landings. There was no evidence of long-term trends in number of strandings in any of the species and their abundances were, therefore, considered to have been relatively stable during the study period.

2.3 Management frameworks (including indicators and targets for MSFD assessments)

WGMME have reported in previous years on the development of common indicators and targets for the Marine Strategy Framework Directive (MSFD) primarily associated with the Marine Atlantic region (e.g. ICES, 2012; 2013; 2014; 2015; 2016; 2017a).

FRANCE: MFSD. The coherent geographic scale for the assessment of Good Ecological Status for marine mammals is the marine region (or subregion). A special effort was therefore made to implement common indicators, mainly within the framework of OSPAR and the advice provided by ICES (2014). These common indicators remain imperfect for a robust assessment of the GES of marine mammal populations due to the lack of available data which limited their calculation during the 2017 Intermediate Assessment of OSPAR. As a result, indicators have been developed at the national level to assess the GES in marine waters under French jurisdiction (and beyond) (Spitz *et al*., 2017). These indicators are summarised in Table 10.

Table 10. French indicators proposed for the 2018 assessment of Good Ecological Status for the marine mammal descriptor (D1). Species are Pp: harbour porpoise, Dd: common dolphin, Sc: striped dolphin, Tt: bottlenose dolphin, Gm: pilot whale, Gg: Risso's dolphin, Ba: minke whale, Bp: fin whale (from Spitz *et al*., 2017)

INDICATORS			
Code	Definition	Marine Units	Species
MM_Capt	Bycatch mortality rate (strandings)	Channel & North Sea + Celtic Seas + Bay of Biscay	Dd, Pp
MM_Abond	Trends in relative abundance of cetaceans	Bay of Biscay	Dd, Sc, Tt, Gm, Gg, Ba
MM_EME	Recurrence of unusual mortality event	Channel & North Sea + Celtic Seas + Bay of Biscay + western Mediterranean Sea	Dd, Pp, Sc
MM_Distri	Trends in occupancy by cetaceans	GdG	Dd, Sc, Tt, Gm, Gg, Ba, Bp

Four marine subregions are assessed with respect to GES under the Biodiversity Descriptor (D1), which includes four functional groups of marine mammals (small odontocetes, deep-diving odontocetes, baleen whales and seals). GES evaluation was hampered by a lack of relevant data on many cetacean species, especially baleen whales and deep-diving odontocetes; and by large data gaps in northwestern Mediterranean Sea. The main limits are data gaps for many species and their habitats, especially offshore ones; and a lack of statistical power to detect with sufficient confidence change in biodiversity indicators. These limitations notwithstanding, GES was not reached in the three subregions (North Sea-Channel, Celtic Seas and Bay of Biscay) where quantitative data were available to inform a partial evaluation. This resulted solely from unsustainable bycatch levels documented on two small odontocete species, the harbour porpoise and the short-beaked common dolphin. Bycatch is the foremost pressure preventing GES with respect to marine mammals in French marine waters: this pressure must be addressed adequately in terms of environmental targets and measures during the next MSFD cycle.

MSFD and GES in the Macaronesian region

Work has also been progressing in the Macaronesian region through the MISTIC SEAS project (http://fundacion-biodiversidad.es/sites/default/files/ mistic-seas-ingles-baja.pdf). This project, funded by the European Commission, focused on the populations of cetaceans, turtles and seabirds with the aim of establishing a common approach and improved coordination in the implementation of MSFD in Portugal and Spain specifically for the Azores, Madeira and the Canary archipelagos. A total of 27 management units (MUs) are proposed for assessing GES in Macaronesian waters. GES indicators and targets have now been developed for cetaceans in this area and are summarised in Table 11. However, there are still insufficient data to develop robust indicators for the abundance and demographic parameters for most cetacean species and many of the proposed indicators will not be made operational until further work is carried out to set baseline values (especially for abundance indicators in Azores and

Canaries) and targets. Further research is also needed to assess if existing monitoring data will enable detecting trends in the proposed indicators.

The only pressure indicator proposed for marine mammals is mortality from ship strikes, applicable only for the sperm whale MU. Still, the sublethal effects of whale-watching and underwater noise in Macaronesia should not be overlooked and there is a need to develop robust indicators to monitor impacts from these activities on a wide range of MUs. Knowledge gaps should be taken into consideration when proposing the Programme of Measures. In addition to including measures to help or maintaining GES, the programme of measures should also list measures needed to make the proposed indicators operational, such as short-term research and pilot studies.

Table 11. GES indicators for the marine mammal in the Macaronesian Region (adapted from Carreira, 2017).

GES indicator and definition	Proposed target	Species	Operationality
D 1.2 population size: The population size does not deviate from the natural fluctuations of the population.	Population size is at or above the baseline levels, with no observed estimated or projected reduction ≥10% over a 20-year period.	*Tursiops truncatus* bottlenose dolphin	Operational in Madeira and Azores (island associated), expected in 2018 for Azores (offshore) and Canaries
		Globicephala macrohynchus Short finned pilot whale	Operational in Madeira, expected in 2018 for Canaries
		Ziphius cavirostris Curvier's beaked whale	Expected in 2018 for Canaries
		Stenella frontalis Atlantic spotted dolphin	Operational in Madeira, expected in 2018 for Azores and Canaries
		Delphinus delphis Short beaked common dolphin	Operational in Madeira
		Grampus griseus Risso's dolphin	expected in 2018 for Azores
		Balaenoptera edeni Bryde's whale	Further development needed for Madeira
		Physeter macrocephalus Sperm whale	expected in 2020 for Azores
D1.2 Population size: Population size attains levels allowing it to qualify to the Least Concern Category of IUCN	Maintain positive population growth rate until GES is reached	*Physeter microcephalus* Sperm whale	Operational in Azores, expected in 2018 for Canaries
		Balaenoptera physalus Fin whale	Expected in 2020 for Azores
D1.3 Population condition: Population demographic characteristics (productivity, survival rate, calf survival, etc.) are	No statistically significant decrease in survival rates from baseline values.	*Globicephala macrohynchus* Short finned pilot whale	Operational in Madeira, expected in 2018 for Canaries
		Grampus griseus Risso's dolphin	expected in 2020 for Azores
		Balaenoptera edeni Bryde's whale	Further development needed for Madeira

GES indicator and definition	Proposed target	Species	Operationality
not adversely affected by human activities and ensure the long-term viability of the population.		*Tursiops truncatus* Bottlenose dolphin	Operational in Madeira and Azores, expected in 2018 for Canaries
		Ziphius cavirostris Curvier's beaked whale	Expected in 2018 for Canaries
		Physeter macrocephalus Sperm whale	Expected in 2020 for Azores.
	Mortality rate from ship strikes is close to zero	*Physeter macrocephalus* Sperm whale	Expected in 2018 for Azores and Canaries, needs further development for Madeira

The 2nd report of MISTIC SEAS I (http://mistic-seas.madeira.gov.pt/en/content/products) proposes a common monitoring approach for the Macaronesia and the MISTIC SEAS II project is underway to conduct a line-transect visual survey in each archipelago (completed in Madeira and Canaries in 2017, planned for summer 2018 in the Azores) to test proposed methodologies and obtain estimates of abundance of proposed MUs and a photo-ID survey for abundance estimation of bottlenose dolphins (to be completed in spring 2018 in the three archipelagos).

Harbour porpoise Assessment Units in the Belt Sea Area (Baltic Sea)

In 2013, WGMME recommended the following Assessment Units (AU) for the harbour porpoise which were largely delineated by ICES areas/division boundaries (Figure 30).

1) North Sea (NS): Area 4, divisions 7.d and part of 3.a (Skagerrak and northern Kattegat), the boundary between NS and Kattegat/Belt Seas is currently being revised (A. Galatius, pers. Comm.);

2) Kattegat and Belt Seas (KBS): Part of Division 3.a (southern Kattegat) and Baltic Areas 22 and 23;

3) Western Scotland and Northern Ireland (WSNI): divisions 6.a, 6.b2;

4) Celtic Sea and Irish Seas (CIS): divisions 7 with the exception of 7.d;

5) Iberian Peninsula (IB): Divisions 8.c and 9.a.

These AUs were reviewed in 2014 and formally submitted to OSPAR as ICES Advice in 2014 (ICES, 2014b). OSPAR adopted these AU for the 2017 Intermediate Assessment (see https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/biodiversity-status/marine-mammals/abundance-distribution-cetaceans/abundance-and-distribution-cetaceans/).

Based on new information as compiled in Sveegaard *et al*. (2015) and the results from SAMBAH (see above), the WGMME proposes the following revised delineations for the North Sea (North Sea) as well as the Kattegat and Belt Seas (KBS) AUs:

1) North Sea (NS): Area 4, divisions 7.d and part of 3.a (Skagerrak and northern Kattegat, north of latitude 56.95°N);

2) Kattegat and Belt Seas (KBS): Part of Division 3.a (southern Kattegat, south of latitude 56.95°N) and Baltic Areas 22, 23 and part of area 24 (west of 13.5°E).

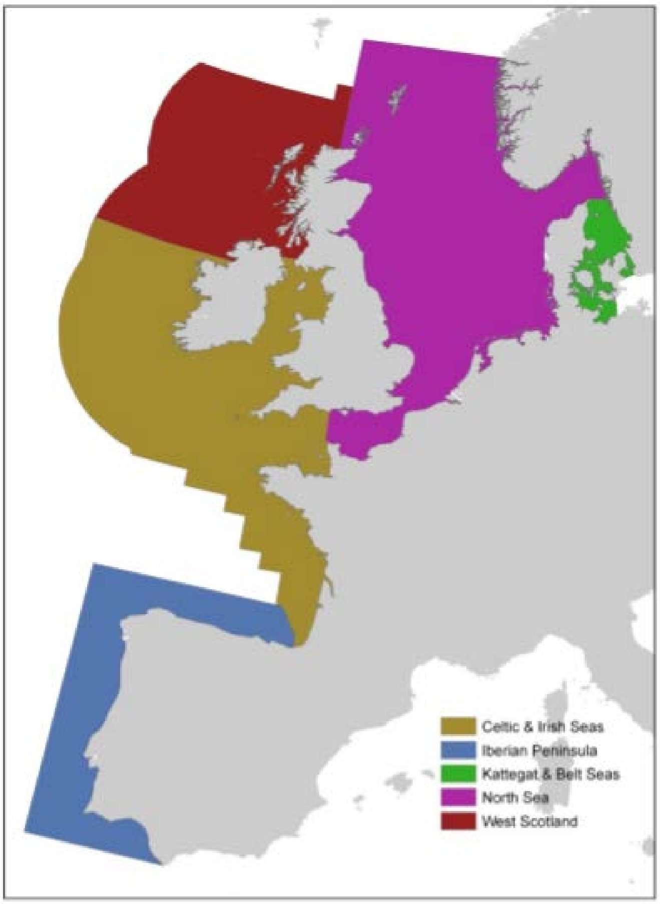

Figure 30. Harbour porpoise assessment units proposed for MSFD indicator assessments. (Taken from ICES, 2014b).

2.4 New information on anthropogenic threats

2.4.1 Fishery bycatch

Please refer to ToR (c).

2.4.2 Pollution: persistent organic pollutants and toxic elements

Desforges *et al*. (2016) reviewed the immunotoxic effects of environmental pollutants in marine mammals in over 50 published reports. Using combined field and laboratory data, Desforges *et al*. (2016) determined effect threshold levels for suppression of lymphocyte proliferation to be between <0.001–10 ppm for PCBs, 0.002–1.3 ppm for Hg, 0.009–0.06 for MeHg, and 0.1–2.4 for cadmium in polar bears and several pinniped and cetacean species. Similarly, thresholds for suppression of phagocytosis were 0.6–1.4 and 0.08–1.9 ppm for PCBs and mercury, respectively. Although data are lacking for many important immune endpoints and mechanisms of specific immune alterations are not well understood, this review revealed a systemic suppression of immune function in marine mammals exposed to environmental contaminants. Exposure to immunotoxic contaminants may have significant population level consequences as a contributing factor to increasing anthropogenic stress in wildlife and infectious disease outbreaks.

SPAIN: Méndez-Fernandez *et al*. (2017) used multivariate analysis to evaluate the ability of PCB patterns to discriminate between five sympatric species: common dolphin, long-finned pilot whale, harbour porpoise, striped dolphin and bottlenose dolphin stranded or bycaught along the Northwest coast of the Iberian Peninsula. The project also aimed to determine which eco-biological factors influence these patterns, thus evaluating the relevance of PCB concentrations as biogeochemical tracers of feeding ecology. Different exposure to PCBs as a consequence of their different dietary preferences or habitats, together with potentially dissimilar metabolic capacities meant the five species could be separated. Sex, age, habitat and the type of prey eaten were the most important eco-biological parameters of those tested. Although, no single congener has been specifically identified as a tracer of feeding ecology, four congeners from the 22 analysed were the most useful and approximately twelve congeners seemed to be enough to achieve good discrimination between the cetaceans studied. Despite more studies are needed, Méndez-Fernandez *et al*. (2017) concluded that PCB patterns can be used as tracers for studying the feeding ecology, sources of contamination and potentially population structure of cetacean species.

FRANCE: In the framework of the MFSD, the observatory Pelagis will establish a monitoring program for the evaluation of organic pollutants and trace elements contamination in marine mammals. As an example, Figure 31 shows the temporal trends of hepatic mercury concentrations and renal cadmium concentrations in harbour porpoises (n = 137). In both cases, concentrations slightly increase during the period, but the most striking feature is the increase of hepatic mercury and renal cadmium concentrations in the outliers over the period.

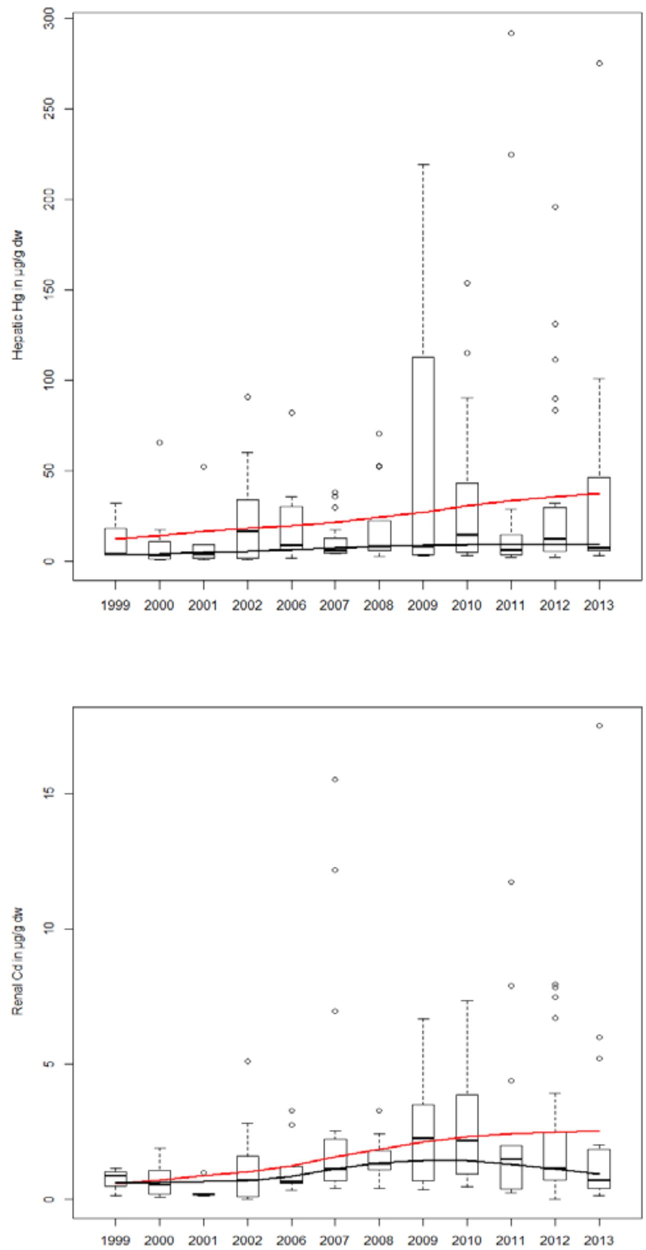

Figure 31. Temporal trends of hepatic mercury concentrations (top) and renal cadmium concentrations (bottom) in 137 individuals of harbour porpoises stranded along the coasts of France (Pelagis data).

2.4.3 Marine debris

GERMANY: Unger *et al*. (2017) studied records of marine debris in and attached to stranded harbour porpoises, harbour seals and grey seals. A total of 6587 carcasses were collected along the German coast between 1990 and 2014; of these the decomposition state allowed for necropsy in 1622 cases (i.e. incl. investigation of the gastro-intestinal tract). Marine debris objects were categorised into fishing related or general debris. General debris includes consumer and industrial debris items. Marine debris items were recorded in 31 carcasses including 14 entanglements (five harbour porpoises, six harbour seals, three grey seals) and 17 cases of ingestion (four harbour porpoises, ten harbour seals, three grey seals). Objects comprised general debris (35.1%)

and fishing related debris (64.9%). Injuries associated with marine debris included lesions, suppurative ulcerative dermatitis, perforation of the digestive tract, abscessation, suppurative peritonitis and septicaemia. This study is the first investigation of marine debris findings in all three marine mammal species from German waters. It demonstrates the health impacts marine debris can have, including severe suffering and death. The results provide needed information on debris burdens in the North and Baltic Seas for implementing management directives, such as the Marine Strategy Framework Directive (MSFD).

NETHERLANDS: Van Franeker *et al.* (2018) studied the frequency of occurrence of plastic other man-made litter in 654 harbour porpoise stomach samples collected in the Netherlands between 2003 and 2013. The frequency of occurrence of plastic litter was 7% with <0.5% additional presence of non-synthetic man-made litter. However, when a dedicated standard protocol for the detection of litter is followed, a considerably higher percentage (15% of 81 harbour porpoise stomachs from the period 2010–2013) contained plastic litter. Results thus strongly depended on methods used and time period considered. Occurrence of litter in the stomach was correlated to the presence of other non-food remains like stones, shells, bog-wood, etc. suggesting that litter was often ingested accidentally when the animals foraged close to the bottom. Most items were small and were not considered to have had a major health impact. No evident differences in ingestion were found between sexes or age groups, with the exception that neonates contained no litter. Polyethylene and polypropylene were the most common plastic types encountered. Compared to earlier literature on the harbour porpoise and related species, results suggest higher levels of ingestion of litter. This is largely due to the lack of dedicated protocols to investigate marine litter ingestion in previous studies. Still, the low frequency of ingestion, and minor number and mass of litter items found in harbour porpoises in the relatively polluted southern North Sea indicates that the species is not a strong candidate for annual monitoring of marine litter trends under the EU marine strategy framework directive.

UK: As highlighted in ICES (2017a), the overall frequency of incidental macroplastic ingestion in cetaceans and pinnipeds is low based on data collected by the UK Cetacean Strandings Investigation Programme during 8200 necropsies. 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. Table 12 provides a summary of the data collected during 2016. There were no impactions, obstructions or pathological change associated with any of the foreign bodies recorded and they were considered to be of incidental significance (Deaville, in press). These included a small fragment of plastic in the first stomach of a sperm whale that stranded at Skegness, England, and in the cardiac stomach of a short-beaked common dolphin that stranded at Kingston Gorse, West Sussex. Two small fragments of soft plastic/cellulose wrap, along with some straw like material and fragments of seaweed were found in the oesophageal and cardiac stomach lumen of a white beaked dolphin, which stranded at Dingieshowe beach in Orkney. A small fragment of paper/debris were found in the fundic stomach of a harbour porpoise that stranded at St Andrews, Fife.

Table 12. Marine debris ingestion or entanglement in cetacean and marine turtle strandings examined at post-mortem in the UK during 2016. ([1]Stomach content data not examined/available). (Taken from Deaville, in press).

Species	PMEs	Marine debris ingestion	Marine debris entanglement
Harbour porpoise	76	1	0
Short-beaked common dolphin	36	1	0
Sperm whale	6[1]	1	0
White beaked dolphin	5	1	0
Striped dolphin	4	0	0
Risso's dolphin	4	0	0
Minke whale	3[1]	0	0
Fin whale	2	0	0
Bottlenose dolphin	1	0	0
Atlantic white-sided dolphin	1	0	0
Sowerby's beaked whale	1	0	0
Pygmy sperm whale	1	0	0
Killer whale	1	0	0
Loggerhead turtle	5	0	0
Leatherback turtle	1	1	0
Kemp's ridley turtle	1	0	0
Green turtle	1	0	0
Total	**149**	**5**	**0**

IRELAND: Lusher *et al*. (2018) reported that 241 of the 2934 stranded cetaceans (at least eleven species) in Ireland presented signs of possible entanglement or interactions with fisheries. Of this number, 52.7% were positively identified as bycatch or as entangled in fisheries items, 26.6% were classified as mutilated and 20.7% could not be related to fisheries, but showed signs of entanglement. Post-mortem examinations were carried out on a total of 528 stranded and bycaught individuals, with 45 (8.5%) having marine debris in their digestive tracts. 21 contained macrodebris, 21 contained microdebris and three had both macro- and microdebris. Forty percent of the ingested debris were fisheries related items. All 21 individuals investigated with the novel method for microplastics contained microplastics, composed of fibres (83.6%) and fragments (16.4%). Deep-diving species presented more incidences of macrodebris ingestion, but it was not possible to investigate this relationship with ecological habitat. More research on the plastic implications to higher trophic level organisms is required to understand the effects of these pollutants.

2.4.4 Underwater noise

DENMARK: van Beest *et al*. (2018) investigated fine-scale movement responses of free-ranging harbour porpoises to capture, tagging and short-term noise pulses from a single airgun. Five porpoises incidentally caught in poundnets in Danish waters were equipped with high-resolution location and dive loggers. All porpoises responded to capture and tagging with longer, faster and more directed movements as well as with shorter, shallower, less wiggly dives immediately after release. Baseline behaviour was resumed in less than or equal to 24 hours after release. After or equal to three days after tagging, the porpoises were exposed to high-intensity noise pulses (2–3 second intervals) for one minute, emitted by a single 10 inch^3 underwater airgun. The porpoises were exposed at ranges of 420–690 m, resulting in estimated sound exposure levels of 135–147 dB re 1 μPa2s. After exposure, two individuals per-formed shorter and shallower dives compared to baseline behaviour, whereof one individual also displayed rapid and directed movements away from the exposure site (a malfunctioning GPS unit precluded any assessment of horizontal parameters for the other one). Noise-induced movement typically lasted for less than or equal to 8 hours with an additional 24 hour recovery period until baseline behaviour was resumed. The remaining three individuals did not show any quantifiable responses to the noise exposure. It is concluded that anthropogenic disturbances may reduce feeding opportunities by changing the porpoises' natural behaviour, and potential population consequences should be a priority research area.

Wisniewska *et al*. (2016) examined foraging interactions by recording high-resolution movement and prey echoes of wild harbour porpoises. Five porpoises incidentally caught in poundnets along the coasts of the Kattegat and Belt Seas were equipped with acoustic and high-resolution movement sensors (three-dimensional acceleration, magnetic field and pressure). Analysis of the 15–23 hour deployments showed that the porpoises made 0–220 foraging attempts per hour during the day and 50–550 attempts per hour at night. The capture success rate was estimated to >90% and the maximum body length of the targeted fish to 3–10 cm, which is smaller than generally found in stomach content analyses of bycaught and stranded harbour porpoises. The high foraging rates support previous findings on the small energetic margins of harbour porpoises and indicate that even moderate disturbance levels may have severe fitness consequences at individual and population levels.

In a comment to Wisniewska *et al*. (2016), Hoekendijk *et al*. (2018) points out that four of the five individuals tagged by Wisniewska *et al*. (2016) were juveniles, and therefore likely fed on smaller fish than the population as a whole. Mainly dependent on smaller prey items, the foraging rate is also likely to be higher than for the population as a whole. Further, Hoekendijk *et al*. (2018) pointed out that when trapped in the poundnet (<24 hours), the porpoises may have starved, also leading to higher foraging rates after release. In a response, Wisniewska *et al*. (2018a) includes data from two additional animals, one adult and one juvenile. The average buzz rare of the two tagged adults is lower than that of the juveniles, but they still appear to target 1500–2000 small fish per day. Further, Wisniewska *et al*. (2018a) explains that there is always fish in the poundnets where the animals are trapped, although it is not known whether the porpoises feed or not during the entrapment. Nevertheless, given that van Beest *et al*. (2018) found that the diving behaviour was affected up to 24 hours after release, the foraging behaviour may also have been so. Both, Hoekendijk *et al*. (2018) and Wisniewska *et al*. (2018a), estimates daily energy intake. Using the average foraging rate for juveniles, a 90% capture success, and a fish size of 1 g, Wisniewska *et al*. (2018a) reach an estimate of roughly 10% of the porpoise' body weight.

Wisniewska *et al*. (2018b) measured vessel noise exposure and foraging efforts of wild harbour porpoises. A total of 19 porpoises incidentally caught in poundnets were equipped with noise sensors (approximately flat frequency response at 0.5–150 kHz) and three-dimensional orientation and pressure sensors. High-quality recordings lasting 12–24 hours were obtained and analysed from seven individuals. The tagged porpoises encountered vessel noise 17–89% of the time. For one animal only low-level vessel noise was recorded, while the others experienced occasional high noise levels associated with vessel passages. Two of the porpoises experienced a passage of a fast ferry, reacting with vigorous fluking, bottom diving and interrupted foraging. The echolocation ceased when the ferry became audible, for one of the porpoises approximately 7 minutes before the point of closest approach and at a distance of 7 km. Regular foraging behaviour resumed 15 minutes after the interruption began. To separate vessel noise from flow noise, a threshold of 96 dB re 1 µPa (16 kHz third-octave) was defined, which is similar to previously suggested threshold for behavioural reactions of porpoises to anthropogenic noise (Tougaard *et al.*, 2015). In the minutes of high vessel noise levels, four of six porpoises produced fewer buzzes with a tendency to longer duration. The exposure time to vessel noise exceeding the threshold for reduced foraging was 0.9–4.3% min of the analysed minutes. It is concluded that frequent vessel exposures may have long-term fitness consequences and concern is raised that other toothed whale species may also be affected.

To test the effectiveness of acoustic harassment devices (AHDs) or 'seal scarers', Mikkelsen *et al*. (2017) exposed harbour porpoises and harbour seals to tone bursts simulating the reduced output of a Lofitech AHD (0.5 s, randomised repetition interval 0.6–90 seconds, 12 kHz, 165 dB re 1 µPa SL peak-peak). At two locations in the inner Danish waters, a loudspeaker was placed 2 m above the seabed (water depths 5–10 m for porpoises and 5–8 m for seals). The animals were localised by a theodolite from a high observation point. The reduced source level was to ensure that animals' responses occurred within the observation range. A total of 12 experiments containing both a control and an exposure period were conducted for porpoises and 13 experiments were conducted for seals. Harbour porpoises were judged to exhibit avoidance reactions out to ranges of 525 m from the sound source, while seals exhibited avoidance out to 100 m. This shows that the two species respond very differently to AHD sounds, which has implications for AHD applications in multispecies habitats. Sound levels required for deterring less sensitive species (seals) can lead to excessive and unwanted large deterrence ranges for more sensitive species (porpoises).

GERMANY/DENMARK: Dähne *et al*. (2017) studied the effects of constructing the DanTysk offshore wind farm by passive acoustic monitoring of pile-driving noise and harbour porpoises. To protect harbour porpoises from hearing loss, a Lofitech seal scarer was used in combination with Aquamark pingers, and bubble curtains were used to attenuate the pile-driving noise. Harbour porpoise echolocation activity was monitored by C-PODs at twelve stations along three transects. At nine of those stations, underwater noise was also recorded. When the seal scarer was engaged, during pile driving and up to five hours after the piling stopped, porpoise echolocation activity was reduced out to 12 km. This is less than the 18–25 km reported from pile driving without bubble curtains, showing that the bubble curtains reduced the temporary habitat loss and risk of hearing loss. The two bubble curtains were more efficient at attenuating the noise when used together (12 dB), than separately (between 7 and 10 dB). The effect was most pronounced above 1 kHz, i.e. at frequencies were porpoises have shown strong behavioural reactions to ship noise components (Dyndo *et al.*, 2015). The findings suggest that noise regulation should be based on frequency-weighted sound levels in addition to broadband levels. The strong reactions to the seal scarer raises concern that it may surpass the reactions to the attenuated pile-driving noise and calls for a re-evaluation of the specifications of seal scarer sounds.

UK: ICES (2017) summarised the current state of knowledge regarding underwater noise from human activities. While policymakers are beginning to address the risk of ecological impact, they are constrained by a lack of data on current and historic levels of both impulsive and continuous noise. Such constraints limit the ability of regulators to assess the potential impact of proposed activities through the Environmental Impact

Assessment (EIA) process and also limits target setting at larger scales. Merchant *et al*. (2017) provides the first nationally coordinated effort to quantify underwater noise levels in order to support UK policy objectives under the EU Marine Strategy Framework Directive (MSFD). Field measurements were made during 2013–2014 at twelve sites around the coast of the UK. For each of the three monitoring regions, four summary metrics of noise level were computed: the mode, median, 90th percentile and the root mean square (RMS) level, which is conventionally used to represent the mean (Table 13). For these metrics, the northern North Sea had the highest noise levels at 125, 250, and 500 Hz, while the southern North Sea had the highest noise levels in the 63 Hz band (due to persistent, localised tonal noise from the nearby power station).

Median noise levels ranged from 81.5–95.5 dB re 1 μPa for one-third octave bands from 63–500 Hz. Noise exposure varied considerably, with little anthropogenic influence at the Celtic Sea site, to several North Sea sites with persistent vessel noise. Comparison of acoustic metrics found that the RMS level was highly skewed by outliers, exceeding the 97th percentile at some frequencies. By contrast, percentile-based metrics and the mode were found to be robust to such outliers. While a number of threshold levels for injury and disturbance have been proposed for acute noise exposure for particular taxa (Popper *et al.*, 2004; Southall *et al.*, 2007; Halvorsen *et al.*, 2012), uncertainty over the effects of noise at the ecosystem scale limits the ability to formulate absolute thresholds for ecologically sustainable noise levels. Until these uncertainties are addressed, progress can be made by establishing monitoring programmes to track levels of noise pollution, and by ensuring that metrics used to describe noise levels are pertinent to assessing the risk of impact to marine life. The current recommendation for the MSFD is to use the RMS level (Dekeling *et al*., 2014), but this metric is strongly influenced by outliers in the distribution (Merchant *et al*., 2012), and so can be skewed away from the general trend in noise levels by a few high amplitudes but unrepresentative events in the time-series. Merchant *et al*. (2017) concluded that environmental indicators of anthropogenic noise should instead use percentiles in order to ensure statistical robustness.

Power analysis indicated that at least three decades of continuous monitoring would be required to detect trends of 1 dB re 1 μ Pa per decade trend. With respect to the ambition of the MSFD to assess and achieve 'Good Environmental Status' with respect to underwater noise by 2020, the results of Merchant *et al*. (2017) imply that the MSFD Indicator for ambient noise should be redefined in terms of absolute noise levels (e.g. thresholds) rather than continuing to use the present trend-based indicator.

Table 13. Summary metrics of noise level for UK monitoring regions, with data for the North Sea sites represented by the median value among the nine monitoring sites. Metrics represent all data from each site (i.e. over both monitoring years where applicable). All metrics are sound pressure levels in the corresponding 1/3-octave frequency band, in units of dB re 1 μ Pa. (Taken from Merchant *et al.*, 2017).

	Monitoring region	63 Hz	125 Hz	250 Hz	500 Hz
Mode	Celtic Sea	75.8	83.2	88.4	91.5
	North Sea	90.0	92.0	94.5	94.3
	Southern North Sea	94.0	87.0	72.7	82.3
Median	Celtic Sea	82.0	83.3	87.1	89.7
	North Sea	90.5	93.6	95.5	94.6
	Southern North Sea	94.7	86.0	78.9	83.5
90^{th} percentile	Celtic Sea	93.2	93.3	96.0	96.9
	North Sea	100.3	103.5	103.9	103.3
	Southern North Sea	102.0	96.5	94.3	93.3
RMS level	Celtic Sea	101.6	102.3	102.9	99.9
	North Sea	101.8	103.8	104.5	104.2
	Southern North Sea	110.8	113.1	113.3	104.9

USA: Forney *et al*. (2017) present five case studies to illustrate the concerns of conventional mitigation approaches focusing on the reduction of direct physical injury from intense anthropogenic noise. Conventional mitigation efforts often attempt to minimize injury by enabling animals to move away from the noise source. However, for species with very high site fidelity, particularly those with very small local populations, displacement is likely to include increased stress and reduced foraging success, with associated effects on survival and reproduction. Further, in some cases, displacement may also increase the risk of e.g. bycatch in nearby fisheries. To provide managers and operators with a more robust means of assessing and avoiding potential, Forney *et al*. (2017) present an expanded framework that covers both displacement and direct effects of intense anthropogenic noise exposure and acknowledges scientific uncertainty. One of the five case studies presented concern the Cuvier's beaked whales *Ziphius cavirostris* off the US Atlantic coast.

2.4.5 Ship strikes

ICES (2017a) noted that population level effects are most likely for small populations and that concerns have been raised for North Atlantic Right Whales (*Eubalaena glacialis*). During 2017, elevated North Atlantic right whale mortalities occurred primarily in Canada. A total of 16 confirmed dead stranded whales, of which twelve occurred in Canada and four in the US, and further five live whale entanglements in Canada. This was declared an unusual mortality event (Figure 32).

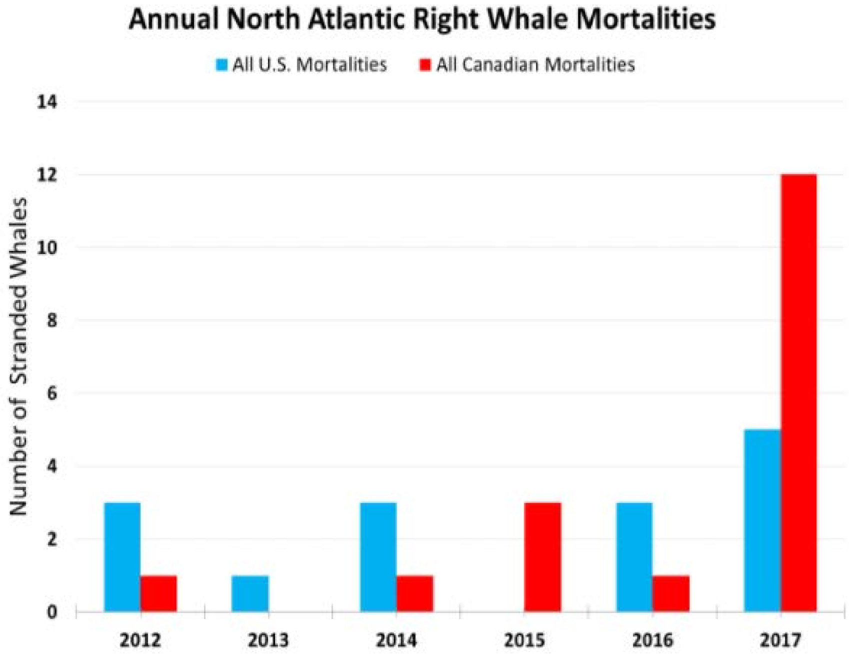

Figure 32. North Atlantic right whale mortalities events between 2012 and 2017 (NOAA; http://www.nmfs.noaa.gov/pr/health/mmume/2017northatlanticrightwhaleume.html) .

Critical habitat for this species was defined and conservation measures were implemented in the USA based on data indicating that most individuals inhabited the Bay of Fundy and Roseway Basin during summer and autumn (Vanderlaan *et al*., 2008). During 2017, the number of individuals using these areas appears to have diminished, with presumably larger numbers moving into the Gulf of St Lawrence (Frasier and Reeves, 2017). These incidents highlight the fact that as species distributions shift, they require dynamic and flexible conservation solutions.

2.4.6 Tourism

The interest in wildlife watching is growing and wildlife tourism activities are currently developed in new and remote areas of the world (e.g. Hoover-Miller *et al*., 2013) or in areas where species are recovering. WGMME has not previously covered this and, therefore, this section will include some more background.

Wildlife tourism can have positive effects on the local economy of rural communities and may facilitate an awareness of wildlife conservation among tourists and stakeholders (Higginbottom, 2004; Sekercioglu, 2002). However, wildlife tourism can affect wild animals of different taxa negatively at an individual or a population level (e.g. Kovacs and Innes, 1990; Whoeler *et al*., 1994; Carney and Sydeman, 1999; Johnson and Lavigne, 1999; Lusseau *et al*., 2006; Creel *et al*., 2002). Disturbance can cause physiological responses and affect the natural behaviour of wildlife (e.g. Carney and Sydeman, 1999; Barja *et al*., 2007; Granquist and Sigurjónsdóttir, 2014), for example causing animals to become more vigilant and could cause fleeing (e.g. Jayakody *et al*., 2008) or other behaviour that causes an increase in energy expenditure (Christiansen *et al*., 2010). Animals may also be forced to spend less time on essential behaviours such as attending their offspring and resting (Carney and Sydeman, 1999; Kovacs and Innes, 1990; Tyler, 1991). Further, tourism related disturbance may lead to abandonment of optimal breeding and resting sites (Johnson and Lavigne, 1999; Whoeler *et al*., 1994). Consequently, effects of tourism may have serious impact on threatened pinniped species (see Kovacs *et al*., 2012) and concerns for negative effects on the welfare of pinniped and cetacean species have increased (Christiansen and Lusseau, 2014).

ICELAND: The interest for marine mammal watching has been growing, with up to 1/3 of tourists visiting Iceland going whale watching (Ferdamamálastofa, 2017). Seal watching sites are also currently being developed in several areas of Iceland. Therefore, effects of land-based (Granquist and Sigurjónsdóttir, 2014) and boat-based (Granquist *et al*., in prep.) seal watching on seal colonies have been studied, along with suitable management actions to reduce potential impacts. Harbour seals were found to be more vigilant when many tourists were visiting a land-based seal watching site and in addition seal distribution was affected (the seals hauled out to a higher extent on skerries further away from land). Tourist behaviour is important; e.g. when tourists behaved in a calmer way, seals were less vigilant. This indicates that modifying of tourist behaviour can reduce negative impact of wildlife tourism on harbour seal colonies (Granquist and Sigurjónsdóttir, 2014). To facilitate mitigation, interdisciplinary management models have been developed (Granquist and Nilsson, 2013; Granquist and Nilsson, 2016) and existing codes of conducts for seal watching investigated (Öqvist *et al*., 2018). Tourist behaviour can be modified through signage, especially if the information was teleological (explaining the background of the behaviour recommendations) (Marschall *et al*., 2016). Further research is underway.

UK: Studies have indicated that in the Cardigan Bay Special Area of Conservation, bottlenose dolphin encounter rates are inversely related to encounter rates with marine wildlife watching vessels (Pierpoint *et al*., 2009). Small boat traffic in the area has been increasing steadily in recent years. In southern parts of Cardigan Bay, there has been a long-term code of conduct in place, introduced by the local authority, and compliance to this has been assessed as high. In the north, there is no code of conduct until very recently, and the profile of vessels differs somewhat, with more yachts and speed craft and less wildlife trip boats and fishing vessels. Intensive visual observations and theodolite tracking indicate that the established code of conduct has encouraged all vessels to stay further away from dolphins, and that such practices are reducing the number of negative responses from dolphins to vessels, even during close encounters or when boat encounter rates are high.

UK: The introduction of the Wildlife and Countryside Act 1981 (and its subsequent amendments, hereafter referred to as WCA) lead to a significant shift in the legal apparatus for nature conservation in the UK. The WCA covers protection of wildlife making it an offence (subject to exceptions) to intentionally kill, injure or take, possess, or trade in any wild animal listed in schedule 5, and prohibits interference with places used for shelter or protection, or intentionally disturbing animals occupying such places. Of the cetacean species utilising UK waters, schedule 5 originally listed bottlenose dolphin, common dolphin and harbour porpoise. Since its introduction, there have been various amendments to the text of the Act and to schedules, with an extension to cover all cetaceans today.

Despite WCA being in place since 1981, the first successful conviction under the amended legislation for disturbance did not occur until 2007. At Banff Sheriff Court, Scotland, a man was found guilty of intentionally or recklessly disturbing or harassing bottlenose dolphins by splashing water in an attempt to attract the attention of the animals and also of driving a jet ski at high speed in a reckless and erratic manner at and around the animals. In complete contrast, in 2013, a bottlenose dolphin calf was killed in a private vessel collision also linked to harassment. A small inshore group of dolphins spent time swimming close to the shore in the Camel estuary near Padstow in Cornwall and up to 25 boats were seen around them. When the pod left the area, the body of a dolphin calf was found. Cornwall Wildlife Trust and the British Divers Marine Life Rescue jointly stated that the death was the result of harassment by vessels. Although photographic evidence was forthcoming, no prosecution was brought because the defendant claimed he had no knowledge that he was committing an offence. Of particular relevance to these very different outcomes is the Nature Conservation (Scotland) Act 2004, which contained provisions for Scottish Natural Heritage (SNH) to create a Scottish Marine Wildlife Watching Code https://www.nature.scot/professional-advice/land-and-sea-management/managing-coasts-and-seas/scottish-marine-wildlife-watching-code). This sets out recommendations, advice and information relating to commercial and leisure activities involving the watching of marine wildlife, clearly articulating everyone's responsibility to adhere to the legal requirements with regard to disturbance and harassment.

It was the existence of this code that in part enabled the disturbance prosecution to be taken forward in Scotland while, what might be considered a far worse infringement of the WCA in England, was not pursued. In contrast to cetaceans, seals receive far less legal protection in the UK. Protection from disturbance and harassment only applies when the seals are hauled out and/or pupping within designated sites. There have been numerous cases where people get too close to the seals (e.g. Figure 33).

Figure 33. Forvie National Nature Reserve, Aberdeenshire (February 2017) (courtesy of Scottish Natural Heritage).

Disturbance and harassment in the marine environment are especially difficult issues to obtain sufficient evidence in order to prove an infringement of the legislation. However, the **WGMME concludes** that existence of statutory rather than voluntary guidelines is needed in future. The Statutory Nature Conservation Bodies have recently proposed development of a code(s) similar to that applying to Scottish waters for other UK marine areas.

2.5 Intersessional workshop

A Workshop on Predator–prey Interactions between grey seals and other marine mammals (WKPIGS) focused on predatory behaviour of grey seals towards other grey seals, harbour seals and harbour porpoises in European waters was convened in April 2017 in response to a recommendation from the ICES, WGMME (2015).

It was attended by 30 scientists from organisations in six nations across Europe, and the USA and aimed to define and harmonise the pathological indicators of grey seal predation events across nations and to collate data on the prevalence and distribution of such events. A further objective was to discuss methods to aid in detection of predation events and potential population-level consequences of reported incidences (ICES, 2017b).

2.6 Recommendations

The WGMME recommends to

1) further investigate the reasons for a decline in harbour and grey seal abundance in Iceland;
2) investigate possible effects of growing human use of the marine environment on the carrying capacity of the areas for seals;
3) re-evaluate the status of ringed seals in the Baltic Sea. The species is listed as 'least concern' in the latest IUCN Red List report (2015), but the subpopulations of the Baltic subspecies are facing a risk of regional extinction;
4) revise Assessment Units for harbour porpoises in the Belt Sea.

2.7 References

Arts, F. A., Lilipaly, S. and Strucker, R.C.W. 2017. Watervogels en zeezoogdieren in de Zoute Delta 2015/2016. RWS Centrale Informatievoorziening BM pp.

Authier, A., Dorémus, G., Van Canneyt, O., Boubert, J.J., Gautier, G., Doray, M., Duhamel, E., Massé, J., Petitgas, P., Ridoux, V. and Spitz, J. Exploring change in the relative abundance of marine megafauna in the Bay of Biscay, 2004–2016. Progress in Oceanography (in press). https://doi.org/10.1016/j.pocean.2017.09.014.

Bailey, H. and Thompson, P. 2010. Effect of oceanographic features on fine-scale foraging movements of bottlenose dolphins. Mar Ecol Prog Ser 418, 223−233

Baker, I., O'Brien, J., McHugh, K., Ingram, S. N. and Berrow, S. 2017. Bottlenose dolphin (*Tursiops truncatus*) social structure in the Shannon Estuary, Ireland, is distinguished by age- and area-related associations. Mar Mam Sci (in press). doi:10.1111/mms.12462.

Balcomb, K.C. III, and Claridge, D. E. 2001. A mass stranding of cetaceans caused by naval sonar in the. Bahamas. Bahamas Journal of Science 8 (2), 2–12.

Barja, I., Silván, G., Rosellini, S., Pineiro, A., Gonzalez-Gil, A., Camacho, L., Illera, J.C. 2007. Stress physiological responses to tourist pressure in a wild population of European pine marten. J. Steroid Biochem. Mol.Biol. 104, 136–142.

Benke, H., Bräger, S., Dähne, M., Gallus, A., Hansen, S., Honnef, C.G., Jabbusch, M., Koblitz, J.C., Krügel, K., Liebschner, A., Narberhaus, I., and Verfuß, UK 2014. Baltic Sea harbour porpoise populations: status and conservation needs derived from recent survey results. Mar. Ecol. Prog. Ser. 495, 275–290.

Börjesson, P., Read, A.J. 2003. Variation in timing of conception between populations of the harbor porpoise. J. Mammal. 84, 948–955.

Bodewes, R., Zohari, S., Krog, J. S., Hall, M. D., Harder, T. C., Bestebroer, T. M., van de Bildt, M. W. G., *et al*. 2016. Spatiotemporal analysis of the genetic diversity of seal influenza A(H10N7) virus, Northwestern Europe. Journal of Virology, 90: JVI.03046-15.

Boisseau, O., A. Moscrop, A. Cucknell, R. McLanaghan and D. Wall. 2011. An acoustic survey for beaked whales in the Rockall Trough. Report to the International Whaling Commission, SC/63/SM2.

Brasseur S, Czeck R, Galatius A, Jensen, LF, Jess A, Korber P, Pund R, Siebert U, Teilmann J, Klöpper S. 2017. TSEG Grey Seal surveys in the Wadden Sea and Helgoland in 2016–2017. CWSS 4 pp. available at http://www.waddensea-secretariat.org/sites/default/files/CWSS_Internal/TMAP/Marine_Mammals/17-06-07_greysealreport2017.pdf .

Brasseur, S. 2018. Stranding and Rehabilitation in Numbers: Population development and stranding data on the Dutch coasts 1990–2016; Analysis of new data from a public database. Den Helder, Wageningen Marine Research, Wageningen Marine Research report C108/17. 36 pp. available from http://edepot.wur.nl/440805.

Breen, P., Brown, S., Reid, D. and Rogan, E. 2017. Where is the risk? Integrating a spatial distribution model and a risk assessment to identify areas of cetacean interaction with fisheries in the northeast Atlantic. Ocean & Coastal Management, 136, 148–155.

Brough TE, Henderson S, Guerra M, and Dawson SM. 2016. Factors influencing heterogeneity in female reproductive success in a critically endangered population of bottlenose dolphins. End Spec Res., 29:255–270.

Carlén I., Thomas, L., Carlström, J., Amundin, M., Teilmann, J., Tregenza, N., Tougaard, T., Koblitz, J.C., Sveegaard, S., Wennerberg, D., Loisa, O., Dähne, M., Brundiers, K., Kosecka, M., Kyhn, L.A., Tiberi Ljungqvist, C., Pawliczka, I., Koza, R., Arciszewski, B., Galatius, A., Jabbusch, M., Laaksonlaita, J., Niemi, J., Gallus, A., Benke, H., Blankett, P., Skóra, K.E., Acevedo-Gutiérrez, A. Spatial and seasonal distribution of harbour porpoises in the Baltic Sea provides basis for effective conservation actions. In review, Biological Conservation.

Carney, K.M., Sydeman, W.J. 1999. A review of human disturbance effects on nesting colonial waterbirds. Waterbirds: Int. J. Waterbird Biol. 22,68–79.

Carreira, G. (main author) 2017. Standard indicators and criteria for the islands of Macaronesia: Reaching common basis for monitoring marine biodiversity in Macaronesia. Technical report 1. Grant Agreement No 110661/2015/712629/SUB/ENVC.2. https://drive.google.com/file/d/0B3tenWrcwitfR2NwVG83cWFiRGs/view .

Christiansen, F., Lusseau, D., Stensland, E., Berggren, P. 2010. Effects of tourist boats on the behaviour of Indo-Pacific bottlenose dolphins off the south coast of Zanzibar. Endang. Species Res. 11, 91–99.

Christiansen, F., Lusseau, D. 2014. Understanding the ecological effects of whale- watching on cetaceans. In: Higham, J., Bejder, L., Williams, R. (Eds.), whale-watching, sustainable tourism and ecological management. Cambridge University Press, Cambridge, pp. 177–192.

Clutton-Brock TH. 1988. Reproductive success: studies of individual variation in contrasting breeding systems. University of Chicago Press: Chicago.

Creel, S., Fox, J.E., Hardy, A., Sands, J., Garrott, B., Peterson, R.O. 2002. Snow-mobile activity and glucocorticoid stress responses in wolves and elk. Conserv. Biol. 16, 809–814.

Currey RJC, Dawson SM, Slooten E, Schneider K, Lusseau D, Boisseau O.J. *et al*. 2008. Survival rates for a declining population of bottlenose dolphins in Doubtful Sound, New Zealand: an information theoretic approach to assessing the role of human impacts. Aquat Conserv., 19:658–670.

Dähne, M., Tougaard, J., Carstensen, J., Rose, A., Nabe-Nielsen, J. 2017. Bubble curtains attenuate noise from offshore wind farm construction and reduce temporary habitat loss for harbour porpoises. Mar. Ecol. Prog. Ser. 580, 221–237.

Dars, C., Peltier, H., Dabin, W., Demaret, F., Dorémus, G., Spitz, J, and Van Canneyt, O. 2017. Annual Report on strandings in France. Marine mammals strandings along the French coasts in 2016. Annual report of the Observatory Pelagis (University of La Rochelle-CNRS), 65 pp.

Deaville, R. (Ed). In press. Final Report for the period 1st January–31st December 2016. Cetacean Stranding Investigation Programme CSIP. (Contract number MB0111).

Dekeling, R. et al. 2014. Monitoring Guidance for Underwater Noise in European Seas. JRC Sci. Policy Rep. EUR 26557 EN, Publ. Off. Eur. Union, Luxemb. (2014).

Desforges, J. W., Sonne, C., Levin, M., Siebert, U., Guise, S. De, and Dietz, R. 2016. Immunotoxic effects of environmental pollutants in marine mammals. Environment International, 86: 126–139.

DeRuiter SL, Southall BL, Calambokidis J, Zimmer WMX, Sadykova D, Falcone EA, Friedlaender AS, Joseph JE, Moretti D, Schorr GS, Thomas L, Tyack PL. 2013. First direct measurements of behavioural responses by Cuvier's beaked whales to mid-frequency active sonar. Biology Letters 9, 20130223.

den Heyer, C.E., Lang, S.L.C., Bowen, W.D., and Hammill, M.O. 2017. Pup Production at Scotian Shelf Grey Seal (*Halichoerus grypus*) colonies in 2016. DFO Can. Sci. Advis. Sec. Res. Doc. 2017/056. v + 34 pp.

Duck, C. and Morris, C. 2013. An aerial survey of harbour seals in Ireland: Part 2: Galway Bay to Carlingford Lough. August-September 2012. Unpublished report to the National Parks & Wildlife Service, Department of Arts, Heritage & the Gaeltacht, Dublin.

Dyndo, M., Wiśniewska, D.M., Rojano-Doñate, L., and Madsen, P.T. 2015. Harbour porpoises react to low levels of high frequency vessel noise. Scientific Reports 5, 11083.

Ferdamamálastofa. 2017. Tourism in Iceland in figures. https://www.ferdamalastofa.is/static/files/ferdamalastofa/Frettamyndir/2017/juli/tourism-in-iceland-2017-9.pdf.

Fernández R, García-Tiscar S, Santos M.B. *et al*. 2011. Stable isotope analysis in two sympatric populations of bottlenose dolphins *Tursiops truncatus*: evidence of resource partitioning? Mar Biol 158, 1043–1055.

Fontaine MC, Tolley KA, Michaux J.R. *et al*. 2010. Genetic and historic evidence for climate-driven population fragmentation in a top cetacean predator: the harbour porpoises in European water. Proc R Soc Biol Sci 277, 2829–2837.

Fontaine, M. C., Thatcher, O., Ray, N., Piry, S., Brownlow, A., Davison, J., Jepson, P. *et al*. 2017. Mixing of porpoise ecotypes in southwestern UK waters revealed by genetic profiling. Royal Society Open Science 4, 160992.

Forney, K.A., Southall, B.L., Slooten, E., Dawson, S., Read, A.J., Baird, R.W., Brownell, R.L.B. 2017. Nowhere to go: noise impact assessments for marine mammal populations with high site fidelity. Endanger. Species Res. 32, 391–413.

Fraiser, T. and Randell, R. 2017. Lessons learned from cetacean tragedies. Science, 358(6369),1396–1396.

Fruet, P.F., Genoves, R.C., Moeller, L.M., Botta, S., and Secchi, E.R. 2015. Using mark–recapture and stranding data to estimate reproductive traits in female bottlenose dolphins (*Tursiops truncatus*) of the Southwestern Atlantic Ocean. Mar Biol., 162, 661–673.

Galatius, A., Brasseur, S., Czeck, R., Jeß, A., Körber, P., Pund, R., Siebert, U., Teilmann, J., Klöpper, S. 2017. Aerial surveys of Harbour Seals in the Wadden Sea in 2017. Common Wadden Sea Secretariat. Available at http://www.waddensea-secretariat.org/sites/default/files/downloads/TMAP_downloads/Seals/17-11-09_harboursealreport2017.pdf.

Gallus, A., Dähne, M., Verfuß, UK, Bräger, S., Adler, S., Siebert, U. and Benke, H. 2012. Use of static passive acoustic monitoring to assess the status of the 'Critically Endangered' Baltic harbour porpoise in German waters. Endangered Species Research, 18, 265–278.

Geelhoed S.C.V., Janinhoff, N., Lagerveld, S., Lehnert, L.S. andVerdaat, J.P. 2018. Marine mammal surveys in Dutch North Sea waters in 2017. Wageningen Marine Research (University & Research Centre), Wageningen Marine Research report.

Goldstein, T., J. a. K. Mazet, F. M. D. Gulland, T. Rowles, J. T. Harvey, S. G. Allen, D. P. King, B. M. Aldridge and J. L. Stott. 2004. The transmission of phocine herpesvirus-1 in rehabilitating and free-ranging Pacific harbor seals (*Phoca vitulina*) in California. Veterinary Microbiology 103:131–141.

Goodman, S.J. 1998. Patterns of extensive genetic differentiation and variation among European harbor seals (*Phoca vitulina vitulina*) revealed using microsatellite DNA polymorphisms. Molecular Biology and Evolution 15(2), 104–118.

Granquist.S. and Nilsson, P.Å. 2013. The Wild North: Network Cooperation for Sustainable Tourism in a fragile Marine Environment in the Arctic Region. In: Müller, D., Lundmark, L. and Lemelin, R. (Eds.), New Issues in Polar Tourism pp. 123–132. Heidelberg: Springer.

Granquist, S.M. and Sigurjónsdóttir, H. 2014. The effect of land based seal watching tourism on the haul-out behaviour of harbour seals (*Phoca vitulina*) in Iceland. Applied Animal Behaviour Science 156, 85–93.

Granquist, S. M., and Nilsson, P.Å. 2016. Who's watching whom? - an interdisciplinary approach to the study of seal-watching tourism in Iceland. Journal of Cleaner Production, 111, 471–478.

Haelters, J., Kerckhof, F., Rumes, B., Potin, M. and Jauniaux, T. 2017. Strandingen en waarnemingen van zeezoogdieren en opmerkelijke vissen in België in 2016 [Strandings and sightings of marine mammals and some remarkable fish species in Belgium in 2016]. Rapport BMM 17/01 – MARECO 17/01. 32 pp.

Haelters, J. 2017. Arctic climate fugitives? LUTRA, 60: 1–3.

Haelters, J., Kerckhof, F., Moreau, K., Potin, M., Doom, M. and Jauniaux, T. 2018a (in press). Strandingen en waarnemingen van zeezoogdieren en opmerkelijke vissen in België in 2017 [Strandings and sightings of marine mammals and some remarkable fish species in Belgium in 2017]. Koninklijk Belgisch Instituut voor Natuurwetenschappen (KBIN), Brussels, Belgium.

Haelters, J., Kerckhof, F., Doom, M., Evans, P. G. H., Van den Neucker, T. and Jauniaux, T. 2018b. New Extralimital Record of a Narwhal (*Monodon monoceros*) in Europe. Aquatic Mammals, 44, 39–50.

Halkka, A. and Tolvanen, P. (Eds.) 2017. The Baltic Ringed Seal – An Arctic Seal in European Waters. WWF Finland report 36. 32 pp. available at https://wwf.fi/mediabank/9825.pdf .

Hammill MO, den Heyer CE, Bowen WD. 2014. Grey Seal Population Trends in Canadian Waters, 1960–2014. Canadian Science Advisory Secretariat (CSAS), Research Document 2014/037.

Hammill, M.O., Gosselin, J-F., and Stenson, G.B. 2017. Pup production of Northwest Atlantic grey seals in the Gulf of St Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc. 2017/043. iv + 14 pp.

Härkönen, T. 2015. *Pusa hispida* ssp. *botnica*. The IUCN Red List of Threatened Species 2015: e.T41673A66991604. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T41673A66991604.en. Downloaded on 20 February 2018.

Härkönen, T., Stenman, O., Jüssi, M., Jüssi, I., Sagitov, R. and Verevkin, M. 1998. Population size and distribution of the Baltic ringed seal (*Phoca hispida botnica*). NAMMCO Sci Publ, 1, 167–180.

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

Härkönen, T., Harding, K., Rasmussen, T.D., Teilmann, J., Dietz, R. 2007. Age- and Sex-specific Mortality Patterns in an Emerging Wildlife Epidemic: the Phocine Distemper in European Harbour Seals. PLoS ONE 2(9): e887.

Halvorsen, M. B., Casper, B. M., Woodley, C. M., Carlson, T. J. and Popper, A. N. 2012. Threshold for onset of injury in Chinook salmon from exposure to impulsive pile driving sounds. PLoS One 7, e38968.

Hammond, P.S., Lacey, C., Gilles, A. Viquerat, S., Borjesson, P., Herr, H., Macleod, K., Ridoux, V., Santos, M.B., Scheidat, M., Teilmann, J., Vingada, J. and Øien, N. 2017. Estimates of cetacean abundance in European Atlantic waters in summer 2016 from the SCANS-III aerial and shipboard surveys. Available at https://synergy.st-andrews.ac.uk/scans3/files/2017/05/SCANS-III-design-based-estimates-2017-05-12-final-revised.pdf.

Hansen, R.G., Boye, T.K., Larsen, R.S., Nielsen, N.H., Tervo, O., Nielsen, R.D., Rasmussen, M.H., Sinding, M.H.S. and Heide-Jørgensen, M.P. 2016. Abundance of whales in east and west Greenland in 2015. SC/23/AEWG/08 for the NAMMCO Scientific Committee.

Hauksson, E., Ólafsson, H.G and Granquist, S.M. 2014. Talning útselskópa úr lofti haustið 2012. [An aerial survey of the Icelandic grey seal population in 2012] Institute of Freshwater Fisheries, VMST/14050.

HELCOM. 2017. Population trends and abundance of seals. HELCOM core indicator report. ISSN 2343–2543.

Henderson SD, Dawson SM, Currey JC, Lusseau D, Schneider K. 2014. Reproduction, birth seasonality, and calf survival of bottlenose dolphins in Doubtful Sound, New Zealand. Mar Mamm Sci., 30:167–180.

Higginbottom, K. 2004. 'Wildlife tourism: An introduction', in Higginbottom, K. (ed.) Wildlife Tourism: Impacts, Management and Planning. Common Ground Publishing Pty Ltd, pp. 1–14.

Hoekendijk, J.P.A., Spitz, J., Read, A.J., Leopold, M.F., Fontaine, M.C. 2018. Resilience of harbor porpoises to anthropogenic disturbance: Must they really feed continuously? Mar. Mamm. Sci. 34, 258–264.

Hoover-Miller, A., Bishop, A., Prewitt, J., Conlon, S., Jezierski, C., Armato,P. 2013. Efficacy of voluntary mitigation in reducing harbour seal disturbance. J. Wildl. Manag. 77, 689–700.

ICES. 2012. Report of the Working Group on Marine Mammal Ecology (WGMME), 5– 8 March 2012, Copenhagen, Denmark. ICES CM 2012/ACOM:27. 146 pp.

ICES. 2013. Report of the Working Group on Marine Mammal Ecology (WGMME), 4– 7 February 2013, Paris, France. ICES CM 2013/ACOM:26. 117 pp.

ICES. 2014. Report of the Working Group on Marine Mammal Ecology (WGMME), 10–13 March 2014, Woods Hole, Massachusetts, USA. ICES CM 2014/ACOM:27. 234 pp.

ICES. 2014b. OSPAR request on implementation of MSFD for marine mammals. In Report of the ICES Advisory Committee, 2014. ICES Advice 2014, Book 1, Section 1.6.6.1

ICES. 2015. Report of the Working Group on Marine Mammal Ecology (WGMME), 9– 12 February 2015, London, UK. ICES CM 2015/ACOM:25. 114 pp.

ICES. 2016. Report of the Working Group on Marine Mammal Ecology (WGMME), 8– 11 February 2016, Madrid, Spain. ICES CM 2016/ACOM:26. 117 pp.

ICES. 2017a. Report of the Working Group on Marine Mammal Ecology (WGMME), 6– 9 February 2017, St Andrews, Scotland, UK. ICES CM 2017/ACOM:27. 103 pp.

ICES. 2017b. Report of the Working Group on Predator–prey Interactions between Grey Seals and other marine mammals (WKPIGS), Middelfart, DK. ICES CM 2017/SSGEPD:18. 26 pp.

IJsseldijk, L.L., Kik, M.J.L. and Gröne, A. 2018. Postmortaal onderzoek van bruinvissen (*Phocoena phocoena*) uit Nederlandse wateren, 2017. Biologische gegevens, gezondheidsstatus en doodsoorzaken. Wettelijke Onderzoekstaken Natuur & Milieu, WUR. WOt-technical report [in Dutch].

Jayakody, S., Sibbald, A.M., Gordon, I.J., Lambin, X. 2008. Red deer *Cervus elephus vigilance* behaviour differs with habitat and type of human disturbance. Wildl. Biol. 14, 81–91.

Jensen, L. F., R. Ejbye-Ernst, T. Y. Michaelsen, A. Jensen, D. M. Hansen, M. E. Nielsen and C. Pertoldi. 2017. Assessing the genetic effects of rehabilitating harbor seals (*Phoca vitulina*) in the Wadden Sea using stochastic simulations. Mammal Research 62:363–372.

Jepson, P.D., Arbelo, M., Deaville, R., Patterson, I. A. P. , Castro, P., Baker, J. R., Degollada, E., Ross, H.M., Herráez, P., Pocknell, A.M., Rodríguez, F., Howie, F.,E., Espinosa, A., Reid, J.R., Jaber, J.R., Martin, V., Cunningham, A.A. and Fernández, A. 2003, Gas-bubble lesions in stranded cetaceans. Nature 425, 575–576.

Johnson, M.W., Lavigne, D.M. 1999. Mass tourism and the Mediterranean Monk Seal – the role of mass tourism in decline and possible future extinction of Europe's most endangered marine mammal, *Monachus monachus*. Monachus Guardian 2, 31–60.

Johnston DW, Westgate AJ, Read A.J. 2005. Effects of finescale oceanographic features on the distribution and movements of harbour porpoises *Phocoena phocoena* in the Bay of Fundy. Mar Ecol Prog Ser 295: 279−293.

Kesselring, T., Viquerat, S., Brehm, R., and Siebert, U. 2017. Coming of age: - Do female harbour porpoises (*Phocoena phocoena*) from the North Sea and Baltic Sea have sufficient time to reproduce in a human influenced environment? PLoS ONE, 12: 1–14.

Kogi K, Hishii T, Imamura A, Iwatani T, Dudzinski KM. 2004. Demographic parameters of Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) around Mikura Island, Japan. Mar Mamm Sci., 20, 510–526.

Kovacs, K.M., Innes, S. 1990. The impact of tourism on harp seals (*Phoca groenlandica*) in the Gulf of St Lawrence, Canada. Appl. Anim. Behav.Sci. 26 (1–2), 15–26.

Kovacs, K.M., Aguilar, A., Aurioles, D., Burkanov, V., Campagna, C., Gales, N., Gelatt, T., Goldsworthy, S.D., Goodman, S.J., Hofmeyr, G.J.G., Härkönen,T., Lowry, L., Lydersen, C., Schipper, J., Sipilä, T., Southwell, C., Stuart, S.,Thompson, D., Trillmich, F. 2012. Global threats to pinnipeds. Marine Mamm. Sci. 28, 414–436.

Krog JS, Hansen MS, Holm E, Hjulsager CK, Chriél M, Pedersen K, Andresen LO, Abildstrøm M, Jensen TH, Larsen LE. 2015. Influenza A(H10N7) virus in dead harbor seals, Denmark. Emerg Infect Dis J 21: 684.

Lah, L., Trense, D., Benke, H., Berggren, P., Gunnlaugsson, Þ., Lockyer, C., Öztürk, A., Öztürk, B., Pawliczka, I., Roos, A., Siebert, U., Skóra, K., Víkingsson, G., Tiedemann, R. 2016. Spatially Explicit Analysis of Genome-Wide SNPs Detects Subtle Population Structure in a Mobile Marine Mammal, the Harbor Porpoise. PLOS ONE 11, e0162792.

Laran, S., Authier, M., Blanck, A., Doremus, G., Falchetto, H., Monestiez, P., Pettex, E., Stephan, E., Van Canneyt. O. and Ridoux, V. 2017. Seasonal distribution and abundance of cetaceans with- in French waters: Part II: The Bay of Biscay and the English Channel. Deep Sea Research II 141: 31–40.

Leeney RH, Amies R, Broderick A.C. *et al*. 2008. Spatio-temporal analysis of cetacean strandings and bycatch in a UK fisheries hotspot. Biodivers Conserv 17:2323–2338.

López, A. 2003. Estatus dos pequenos cetáceos da plataforma de Galicia. PhD Thesis, University of Santiago de Compostela, Spain.

López, A., Santos, M.B., Pierce, G.J. *et al*. 2002. Trends in strandings and by-catch of marine mammals in north-west Spain during the 1990s. J Mar Biol Assoc UK 82:513–521.

Lockyer, C., Kinze, C. 2003. Status, ecology and life history of harbour porpoise (*Phocoena phocoena*), in Danish waters. NAMMCO Sci. Publ. 5, 143–175.

Lohrengel, K., Evans, P.G.H., Lindenbaum, C.P., Morris, C.W., Stringell, T.B. 2017. Bottlenose dolphin monitoring in Cardigan Bay 2014–2016, NRW Evidence Report No: 191, 163 pp, Natural Resources Wales, Bangor.

Lusher, A.L., Hernandez-Milian, G., Berrow, S., Rogan, E., and O'Connor, I. 2018. Incidence of marine debris in cetaceans stranded and bycaught in Ireland: Recent findings and a review of historical knowledge. Environmental Pollution, 232, 467–476.

Lusseau, D., Slooten, E., Currey, R.J.C. 2006. Unsustainable dolphin-watching tourism in Fiordland, New Zealand. Tour. Mar. Environ. 3, 173–178.

Mann J, Connor RC, Barre LM, Heithaus M.R. 2000. Female reproductive success in bottlenose dolphins (*Tursiops truncatus*): life history, habitat, provisioning and group-size effects. Behav Ecol., 11: 210– 219.

Mannocci L, Roberts JJ, Miller DL, Halpin PN. 2017. Extrapolating cetacean densities beyond surveyed regions to qualitatively assess human impacts on populations in the high seas. Conservation Biology, 31, 601–614.

Marschall, S., Graquist, S.M. and Burns, G.L. 2016. Interpretation in Wildlife Tourism: Assessing the effectiveness of signage to modify visitor behaviour at a seal watching site in Iceland. Journal for outdoor recreation 17, 11–19.

Méndez-Fernandez, P., Simon-Bouhet, B., Bustamante, P., Chouvelon, T., Ferreira, M., López, A., Moffat, C.F., Pierce, G.J., Russell, M., Santos, M.B., Spitz, J., Vingada, J.V., Webster, L., Read, F.L., González, A.F., and Caurant, F. 2017. Inter-species differences in polychlorinated biphenyls patterns from five sympatric species of odontocetes: Can PCBs be used as tracers of feeding ecology? Ecological Indicators, 74, 98–108.

Méndez-Fernandez P, Webster L, Chouvelon T. *et al*. 2014. An assessment of contaminant concentrations in toothed whale species of the NW Iberian Peninsula: part I. Persistent organic pollutants. Sci Total Environ 484:196–205.

Merchant, N. D., Blondel, P., Dakin, D. T. and Dorocicz, J. 2012. Averaging underwater noise levels for environmental assessment of shipping. J. Acoust. Soc. Am. 132, EL343–EL349.

Mikkelsen, L., Rigét, F.F., Kyhn, L.A., Sveegaard, S., Dietz, R., Tougaard, J., Carlström, J.A., Carlén, I., Koblitz, J.C., Teilmann, J. 2016. Comparing distribution of harbour porpoises (*Phocoena phocoena*) derived from satellite telemetry and passive acoustic monitoring. PloS One 11, e0158788.

Mikkelsen, L., Hermannsen, L., Beedholm, K., Madsen, P.T., Tougaard, J. 2017. Simulated seal scarer sounds scare porpoises, but not seals: species-specific responses to 12 kHz deterrence sounds. R. Soc. Open Sci. 4, 170286.

Murphy, S., Pinn, E.H. and Jepson, P.D. 2013. The short-beaked common dolphin (*Delphinus delphis*) in the North-East Atlantic: distribution, ecology, management and conservation status. Oceanogr Mar Biol An Annu Rev 51, 193–280.

NAMMCO. 2018. Report of the workshop "Cetacean abundance and distribution in the North Atlantic". 28 and 29 October 2017, Halifax, Canada.

Nilssen, K.T. and Bjørge, A. 2017a. Havert og steinkobbe [Grey and harbour seals]. Pages 68–69 in I.E. Bakketeig, M. Hauge & C. Kvamme (eds). Havforskningsrapporten 2017. Fisken og havet, særnr, 1-2017. 98 pp.

Nilssen, K.T. and Bjørge, A. 2017b. Status for kystsel. Anbefaling av jaktkvoter for 2018 [Status for coastal seals. Recommendation for harvest quotas for 2018]. Document to the Norwegian Marine Mammal Scientific Advisory Board, October 2017. 9 pp.

Nordström, M., Högmander, J., Halkka, A., Keränen, S., Kunnasranta, M., Nummelin, J., Miettinen, M., Niinimäki, T. and Tolvanen, P. 2011. Itämerennorppa Saaristomerellä – Unohdettu uhanalainen. – Maailman luonnon säätiön WWF Suomen rahaston raportteja 28.

Ó Cadhla, O., Tara Keena, Denis Strong, Callan Duck, Lex Hiby. 2013. Monitoring of the breeding population of grey seals in Ireland, 2009–2012. National Parks and Wildlife Service, Department of the Environment, Heritage and the Gaeltacht, Dublin, Ireland.

Öqvist, E.L., Granquist, S.M., Burns, G.L. and Angerbjörn, A. 2018. Seal watching: an evaluation of codes of conduct. Tourism in marine environments. https://doi.org/10.3727/154427317X14964473293699.

Oksanen, S.M., Niemi, M., Ahola, M.P. and Kunnasranta, M. 2015. Identifying foraging habitats of Baltic ringed seals using movement data. Movement ecology, 3(1), 33.

Olsen, M. T., Wesley Andersen, L., Dietz, R., Teilmann, J., Harkonen, T., Siegismund, H.R. 2014. Integrating genetic data and population viability analyses for the identification of harbour seal (*Phoca vitulina*) populations and management units. Molecular Ecology 23: 815–831.

Oudejans, M. G. 2014. Cetacean monitoring undertaken during the Blue Whiting Acoustic Survey (BWAS) 2014. Report to the National Parks and Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Dublin, Ireland. Dulra Research, Belmullet, County Mayo, Ireland. 16 pp.

Palmer, K.J., Brookes, K., Rendell, R. 2017. Categorizing click trains to increase taxonomic precision in echolocation click loggers. J. Acoust. Soc.Am. 142(2), 863–877.

Peltier, H., Van Canneyt, O., Dabin, W., Dars, C., Demaret, F. and Ridoux, V. 2017. New Fishery-related Unusual Mortality and Stranding Events of Common Dolphins in the Bay of Biscay, February–March 2017, France. Working Paper of the IWC SC, HIMWP8, 6 p.

Pierce GJ, Santos MB, Murphy S. *et al*. 2008. Bioaccumulation of persistent organic pollutants in female common dolphins (*Delphinus delphis*) and harbour porpoises (*Phocoena phocoena*) from western European seas: geographical trends, causal factors and effects on reproduction and mortality. Environ Pollut 153:401–415.

Pierpoint, C. 2008. Harbour porpoise (*Phocoena phocoena*) foraging strategy at a high energy, near-shore site in south-west Wales, UK. J Mar Biol Assoc UK88: 1167−117.

Pierpoint, C., Allan, L., Arnold, H., Evans, P., Perry, S., Wilberforce, L., and Baxter, J. 2009. Monitoring important coastal sites for bottlenose dolphin in Cardigan Bay, UK. Journal of the Marine Biological Association of the United Kingdom, 89: 1033–1043.

Popper, A. N. *et al*. 2014.. ASA S3/SC1.4 TR-2014 Sound Exposure Guidelines for Fishes and Sea Turtles: A Technical Report prepared by ANSI-Accredited Standards committee S3/SC1 and registered with ANSI., doi: 10.1007/978-3-319-06659-2 (American National Standards Institute, 2014).

Raateoja, M. and Setälä, O. (Eds.) 2016. The Gulf of Finland assessment. Reports of the Finnish Environmental Institute, 26, 363 pp.

Robinson KP, Sim TMC, Culloch RM, Bean TS, CordobaAguilar I, Eisfeld S.M. *et al*. 2017. Female reproductive success and calf survival in a North Sea coastal bottlenose dolphin (*Tursiops truncatus*) population. PLoS ONE 12(9): e0185000.

Rogan, E., and G. Hernandez-Milian. 2011. Preliminary analysis of beaked whale strandings in Ireland: 1800–2009. Report to the International Whaling Commission SC/63/SM 19.

Rogan, E., A. Canadas, K. Macleod. *et al*. 2017. Distribution, abundance and habitat use of deep diving cetaceans in the North-East Atlantic. Deep Sea Research Part II. 141, 8–19.

Roos, A., Neimanis, A. and Ågren, E. 2017. Tumlare 2016 - Hälsostatus och dödsorsaker hos insamlade djur. Naturhistoriska riksmuseet, Enheten för miljöövervakning och forskning, Rapport nr 3: 2017. 6 pp [in Swedish].

Roos, A., Neimanis, A. and Ågren, E. 2018. Tumlare 2017 - Hälsostatus och dödsorsaker hos insamlade djur. Naturhistoriska riksmuseet, Enheten för miljöövervakning och forskning, Rapport nr 2: 2018. 7 pp [in Swedish].

Saavedra, C., Pierce, G.J., Gago, J. *et al*. 2017. Factors driving patterns and trends in strandings of small cetaceans. Mar Biol, 164: 165.

Santos MB, Saavedra C, Pierce G.J. 2014. Quantifying the predation on sardine and hake by cetaceans in the Atlantic waters of the Iberian Peninsula. Deep Sea Res Part II Top Stud Oceanogr 106:232–244.

SCOS. 2017. Scientific Advice on Matters Related to the Management of Seal Populations: 2017. St Andrews, UK, Sea Mammal Research Unit, University of St Andrews: 1–144.

Sekercioglu, C. H. 2002. Impacts of birdwatching on human and avian communities. Environmental Conservation. Cambridge University Press, 29(3), pp. 282–289. doi: 10.1017/S0376892902000206.

Siebert U, Gilles A, Lucke K. *et al*. 2006 A decade of harbour porpoise occurrence in German waters-analyses of aerial surveys, incidental sightings and strandings. J Sea Res 56:65–80.

Smolker, R. A., A. F. Richards, R. C. Connor and J. W. Pepper. 1992. Sex differences in patterns of association among Indian Ocean bottlenose dolphins. Behaviour 123:38–69.

Southall, B. *et al*. 2007. Marine mammal noise-exposure criteria: initial scientific recommendations. Aquat. Mamm. 33, 411–521 (2007).

Spitz, J., Ridoux, V., Trites ,A.W., Laran, S. and Authier, M. Prey consumption by cetaceans reveals the importance of energy-rich food webs in the Bay of Biscay. Progress in Oceanography (in press). http://dx.doi.org/10.1016/j.pocean.2017.09.013.

Spitz, J., Peltier, H. and Authier, M. 2017. Évaluation du Bon État Écologique – Descripteur D1, composante «Mammifères Marins». Rapport scientifique de l'Observatoire PELAGIS – UMS 3462, Université de La Rochelle / CNRS, 160 pp.

Stamper, M. A., F. M. D. Gulland and T. Spraker. 1998. Leptospirosis in rehabilitated Pacific harbor seals from California. Journal of Wildlife Diseases 34:407–410.

Steiner A, Bossley M. 2008. Some reproductive parameters of an estuarine population of Indo-Pacific bottlenose dolphins (*Tursiops aduncus*). Aquat Mamm., 34:84–92.

Sveegaard, S., Galatius, A., Dietz, R., Kyhn, L., Koblitz, J.C., Amundin, M., Nabe-Nielsen, J., Sinding, M.-H.S., Andersen, L.W., Teilmann, J. 2015. Defining management units for cetaceans by combining genetics, morphology, acoustics and satellite tracking. Glob. Ecol. Conserv. 3, 839–850.

Thorbjörnsson, J.G., Hauksson, E., Sigurðsson, G.M. and Granquist, S.M. 2017. Aerial census of the Icelandic harbour seal (*Phoca vitulina*) population in 2016: Population estimate, trends and current status [Landselstalning 2016: Stofnstærðarmat, sveiflur og ástand stofns]. Marine and Freshwater Research Institution, HV 2017-009. Reykjavík 2017. 22 pp.

Tougaard, J., Wright, A.J., Madsen, P.T. 2015. Cetacean noise criteria revisited in the light of proposed exposure limits for harbour porpoises. Mar. Pollut. Bull. 90, 196–208.

Truchon M-H, L'Hérault V, Brêthes J-C. *et al*. 2013. Marine mammal strandings and environmental changes: a 15-year study in the St Lawrence Ecosystem. PLoS ONE 8:59311.

Tyack PL, Zimmer WMX, Moretti D, Southall BL, Claridge D.E. *et al*. 2011. Beaked whales respond to simulated and actual Navy sonar. PLoS ONE 6(3): e17009.

Tyler, N.J.C. 1991. Short-term behavioural responses of Svalbard reindeer *Rangifer tarandus platyrhynchus* to direct provocation by a snowmobile. Biol. Conserv. 56, 179–194.

Vanderlaan, A.S.M., Taggart, C.T., Serdynska, A.R., Kenney, R.D. and Brown, M.W. 2008. Reducing the risk of lethal encounters: vessels and right whales in the Bay of Fundy and on the Scotian Shelf. Endang. Species Res. 4, 283–297.

Vincent, C. *et al*. (in revision); Parc naturel marin d'Iroise, ONCFS, Réserve naturelle des sept iles, Bretagne Vivante, Syndicat Mixte Baie du Mont-Saint-Michel, Réserve naturelle nationale du Domaine de Beauguillot, GMN, Picardie Nature, ADN, GDEAM62, CMNF.

Wall, D., C. Murray, J. O'Brien, *et al*. 2013. Atlas of the distribution and relative abundance of marine mammals in Irish offshore waters 2005–2011. Irish Whale and Dolphin Group, Kilrush, County Clare, Ireland. 58 pp.

Waring, G.T., M.C. Rossman and F.W. Wenzel. 2015. Serious injury determinations for small cetaceans and pinnipeds caught in commercial fisheries off the northeast US coast, 2012. Northeast Fish.Sci. Cent. Ref. Doc. 15-12. 19 pp.

Wells RS, Scott MD. 1990. Estimating bottlenose dolphin population parameters from individual identification and capture-release techniques. In: Hammond PS, Mizroch SA, Donovan GP, editors. Individual recognition of cetaceans: use of photo-identification and other techniques to estimate population parameters. Report of the International Whaling Commission (Special Issue 12). International Whaling Commission: Cambridge; 1990. pp. 407–415.

Wells RS, Scott MD, Irvine AB. 1987. The social structure of free-ranging bottlenose dolphins. In: Genoways HH, editor. Current Mammalogy Volume 1. Plenum Press: New York and London. pp. 247–305.

Whitehead, H. 2008. Analyzing animal societies: Quantitative methods for vertebrate social analysis. The University of Chicago Press, Chicago, IL.

Whitehead, H. 2009. SOCPROG programs: Analysing animal social structures. Behavioral Ecology and Sociobiology 63:765–778.

Whitehead H and Mann J. 2000. Female reproductive strategies of cetaceans, life history and calf care. In: Mann J, Connor RC, Tyack PL, Whitehead H, editors. Cetacean Societies: Field Studies of Whale and Dolphins. University of Chicago Press: Chicago and London; 2000. pp. 217–246.

Whoeler, E.J., Peney, R.L., Creet, S.M., Burton, H.R. 1994. Impacts of human visitors on breeding success and long-term population trends in Adélie Penguins at Casey, Antarctica. Polar Biol. 14, 269–274.

Williamson, L. D., Brookes, K.L., Scott, B.E., Graham, I.M., Bradbury, G., Hammond, P.S. and Thompson, P.M. 2016. Echolocation detections and digital video surveys provide reliable estimates of the relative density of harbour porpoises. Methods in Ecology and Evolution, 7: 762–769.

Williamson, L.D., Brookes, K.L., Scott, B.E., Graham, I.M. and Thompson, P.M. 2017. Diurnal variation in harbour porpoise detection–potential implications for management. Marine Ecology Progress Series, 570, 223–232.

Wisniewska, D.M., Johnson, M., Teilmann, J., Rojano-Doñate, L., Shearer, J., Sveegaard, S., Miller, L.A., Siebert, U., Madsen, P.T. 2016. Ultra-high foraging rates of harbor porpoises make them vulnerable to anthropogenic disturbance. Curr. Biol. 26, 1441–1446.

Wisniewska, D.M., Johnson, M., Teilmann, J., Rojano-Doñate, L., Shearer, J., Sveegaard, S., Miller, L.A., Siebert, U., Madsen, P.T. 2018a. Response to "Resilience of harbor porpoises to anthropogenic disturbance: Must they really feed continuously?" Mar. Mammal Sci. 34, 265–270.

Wisniewska, D.M., Johnson, M., Teilmann, J., Siebert, U., Galatius, A., Dietz, R., Madsen, P.T. 2018b. High rates of vessel noise disrupt foraging in wild harbour porpoises (*Phocoena phocoena*). Proc R Soc B 285, 20172314.

Zohari, S., Neimanis, A., Härkönen, T., Moraeus, C., Valarcher, J.F. 2014. Avian influenza A(H10N7) virus involvement in mass mortality of harbour seals (*Phoca vitulina*) in Sweden, March through October 2014. Euro Surveill: 19(46): 20967.

3 ToR B. Review current issues in relation to indirect impacts of seals on fisheries

3.1 Introduction

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. ICES WGBYC addresses fishery bycatch of marine mammals annually. WGMME has previously reviewed interactions between grey seals and other marine mammals (2016), multispecies models that incorporate marine mammal consumption to assess marine mammal impacts on fishery resources (2015), the prey of marine mammals (2004; 2006), interactions of common dolphins and fisheries (2005) and population and ecosystem impacts of seal removal programmes (2004). In 2003, WGMME discussed the construction of time-series of marine mammal abundance, diet, and consumption rates for the North Sea since 1963 but noted the lack of relevant data.

Indirect impacts of seals (and other marine mammals, notably cetaceans) on fisheries include those arising from resource competition. In ecological theory, two species which use common resources are considered to be competitors if a reduction in the abundance of one (or a decrease in its consumption of the resource) leads to an increase in abundance of the other and vice versa. This is notoriously difficult to demonstrate, especially in marine ecosystems. Experimental studies are necessarily rare (although fishery-induced changes in fish abundance could be viewed as unplanned experiments), comparative studies are limited due to the difficulty of identifying comparable areas or time periods and model-based studies often rely on untestable assumptions. Thus, many studies focus on circumstantial evidence (e.g. extensive overlap in the species and size classes of resources used, large amounts of the resource are removed by one species).

Seal fishery interactions are reported (at least anecdotally) essentially wherever seals are found and, as such, they probably involve all pinniped species present in the ICES region. Marine mammal-fishery interactions have been extensively reviewed in the past (e.g. Northridge, 1984; 1991; Read, 2008; Moore *et al*., 2009). Rather than repeat existing work, the present brief review focuses on the current situation in the North Atlantic, drawing on information published during the last decade as well as current and recent unpublished work.

We focus on two main issues, competition for food and transmission of nematode parasites, which, beside other zoonotic parasites such as the trematode *Pseudamphistomum truncatum*, which is newly emerging in Baltic grey seals (Neimanis *et al*., 2016, Näreaho *et al*., 2017), are the most important parasites transmitted by fish and affecting human health, as well as briefly summarising current understanding of the wider ecological consequences of indirect interactions between seals and fisheries. We also include some comparative information on cetaceans. Several related issues are therefore not covered here, including indirect effects of fisheries on seals, the effect of ghost fishing (e.g. Unger *et al*., 2017), although this is arguably more a direct interaction between fishing gear and marine mammals. There is also a need to further examine the emerging issue of trematode gastric parasites and the role of fish in the life cycles of respiratory parasites of marine mammals. We will therefore suggest a related term of reference for 2019.

3.2 Food consumption by seals in the North Atlantic and resource competition with fisheries

Until about a decade ago, seal population sizes in most of the ICES area were relatively small and competition with fishermen was considered limited to local problems of for example depredation and bycatches (Kauppinen *et al.*, 2005; Westerberg *et al.*, 2006). As hunting and issues such as disturbance and pollution have been regulated at some level, many seal populations have continued to grow and have successfully recovered in some regions. However, some seal populations are currently in decline. ToR A (this report) provides a detailed overview of trends in seal populations. As a result of the recovery in some regions, concerns related to possible competition with fisheries have become more prominent. In addition, there is a growing interest and objective of implementing a more ecosystem-based approach in the management of marine resources, e.g. ecosystem-based fisheries management, in which information on diet of predators such as seals would be important.

Estimates of seal population sizes are sometimes used to estimate annual consumption of fish by the seals (carried out by laymen as well as scientists, with references given later in the text), and as the levels and significance of these estimates grow, and sometimes equal or exceed local fishery landings, they give rise to discussions on regulation of seals. This conflicts with the current public sentiment in favour of saving and protecting marine mammals. Moreover, the harbour, grey and ringed seal are listed as protected species in Annexes II and V of the EU Habitats Directive (Council Directive 92/43/EEC of 21 May, 1992) which means that a) core areas of their habitat must be protected under the Natura 2000 Network and the sites managed in accordance with the ecological requirements of the species (Annex II) and b) it has to be ensured that their exploitation in the wild is compatible with maintaining them in a favourable conservation status (Annex V).

These conflicting points of view and the potential political consequences emphasise the importance of offering correct and complete information on resource use of both seals and fisheries, and to support sustainable solutions in possible conflict areas. This section of the report focuses on studies on seal diet, prey consumption and assessments of impacts of predation by seals on fish stocks. To limit the scope of the current review, the section does not include studies looking at habitat selection and movement ecology, which may also provide crucial information on overlap between seals, fish stocks and fisheries. Furthermore, material on cetaceans and seals outside the North Atlantic, which may include additional fundamental knowledge, has been omitted to reduce the extent of the present review.

3.2.1 Diet

Adequate information on diet is a prerequisite to be able to assess the impact of prey consumption by seals on fish stocks and fisheries and, to be able to understand the effects of fisheries on seal populations. A compilation of studies on diet and prey consumption of seals in the ICES region from 2006 on can be found in Appendix X. A similar assessment, reviewing studies until 2005, was carried out by WGMME in 2006 (ICES, 2006). Different studies may present diet information in various ways, each with its own advantages and disadvantages. These differences need to be taken into account when attempting to compare results from different studies or trying to answer a specific question e.g. concerning spatial and/or temporal variability of prey consumption. ICES (2006) reviewed different methods for marine mammal diet studies; thus, a review of methods is not included here. Nevertheless, it is important to note that different methods used to collect samples and to extract dietary information from these have their own inherent strengths and weaknesses. In addition, to be able to quantify consumption, individual prey consumption rates are scaled up to the population level. Besides information on size and distribution of the seal population, as well as on composition (age; sex), individual energetic requirements and assimilation efficiency, diet data on relative quantities of each prey species eaten are required. For assessments of resource competition, impact on fish stocks and multispecies trophic interactions, it is necessary to have data on size or age distribution of prey species in the diet, together with information on dietary preferences and size and composition of prey stocks, to be able to e.g. study how consumption and predation mortality respond to changes in prey availability (Smout *et al.*, 2014). Information on spatial and temporal patterns of prey consumption is also crucial information. Finally, quantifying resource overlap is still one step short of demonstrating resource competition. In order to show that this is occurring we need to be able to show that increasing removal of fish by fisheries harms seal populations and/or vice versa.

Seals consume a wide range of prey in the North Atlantic, including fish and invertebrates, with occasional events of mammals and birds in the diet. Recently, there have been reports of predation by grey seals on other pinnipeds and on porpoises. The WKPIGS report (ICES, 2017b) notes that around 737 cases of presumed grey seal predation on pinnipeds and porpoises have been recorded between 1985 and 2016. Studies have concluded that in some cases, the contribution of commercially important species, such as cod, herring and salmon can be substantial in the seal diet, although prey preferences vary not only between seal species, regions, seasons and years, but also among individuals, related to e.g. sex, reproduction status and age.

3.2.2 Prey consumption

To estimate prey requirements of (or prey removal by) a seal population, diet data need to be converted to biomass consumption. Quantification of prey consumption is usually based on estimates of individual consumption rates scaled up to the population level, using bioenergetic models of different degrees of complexity. The amount of a specific prey species removed, or the amount of prey required, by a seal population depends on the size, the energy requirement and the prey choices of the population. The energy requirement can be combined with data on diet composition and prey-species energy contents to estimate the biomass consumption of various prey species. All above-mentioned variables may vary in space and time as well as within the population, implying that the quantities of prey consumed, or required, by a seal population can vary substantially e.g. intra-seasonally or depending on the prey species composition of the diet. Estimates of individual energy requirements are usually based on observations or bioenergetic experiments with captive animals (Kastelein *et al.*, 2005; Sparling and Fedak, 2004), bioenergetics measurements of wild animals (Lydersen *et al.*, 1995; Coltman *et al.*, 1997; Jeanniard-du-Dot *et al.*, 2017) or functions of known thermodynamic and biological parameters (Mohn and Bowen, 1996; Hammill and Stenson, 2000). Although some studies differ significantly in their bioenergetics estimates and the individual energy needs can vary a lot between as well as within seal species (Hedd *et al.*, 1997; Reilly *et al.*, 1996; Ryg and Øritsland, 1991; Sparling *et al.*, 2008; Innes *et al.*, 1987), many recent studies often refer to a few published studies of estimates of average individual energy requirement or biomass consumption (Sparling and Smout, 2003; Härkönen and Heide-Jørgensen, 1991; application to cetaceans: e.g. Santos *et al.*, 2014; Andreasen *et al.*, 2017.)

3.3 Seal consumption in an ecological context/competition for food

Utilisation of a common food resource can lead to exploitive competition if the resource is limited in supply and one competitor outcompetes and negatively affects another competitor using the same resource, but competition can also be in the form of interference e.g. by redistribution of prey (Garrison and Link, 2000). Competition can however also be indirect and more complex as the results of predator–prey dynamics in the foodweb, for instance when a seal population feed on a prey or a predator of a commercial fish species (Punt and Butterworth, 1995) or even competition for lower trophic levels in the foodweb (Trites *et al.*, 1997).

Further, indirect interactions may lead to direct interactions, e.g. when seals drown in fishing gear or are injured by contact with active or discarded gear during their search for food (e.g. ICES, 2017a); see Figure 1 below.

Figure 1. Harbour seal drowned in fishing gear and with fishhooks internally from contact with fishing gear (by Jan Haelters (top) and Piet De Laender (bottom)). Photos supplied by Jan Haelters.

3.3.1 Quantification of fish removal by seals

One way of looking at the potential conflicts associated with possible food-resource competition between seals and fisheries is to compare the estimated quantities of the common resource, usually fish, removed by each of the competitors. The purpose can be to present sound data and prevent excessive speculations, sometimes with the rationale that if the fish consumption by a seal population is in the same order of magnitude as fishery catches and it is concluded that the fishery impacts the fish stock, possible impacts by seal predation should be considered too (Hansson *et al.*, 2017). A number of studies have estimated the amount of fish removed by seals, some of which have put seal consumption into perspective by comparing it with fish stock size estimates (Hammond and Grellier, 2006; Hammond and Harris, 2006; Cook and Trijoulet, 2016), fishery catches (Lundström *et al.*, 2014; Lundström *et al.*, 2012; Florin *et al.*, 2013; Hammond and Grellier, 2006; Hammond and Harris, 2006; Vincent *et al.*, 2016; Houle *et al.*, 2016; Planque *et al*., in prep.) and even with other sources of removal such as fish, birds and other mammals (Hansson *et al.*, 2017; Overholtz and Link, 2007; Kaschner *et al.*, 2001; Ruzicka *et al.*, 2013; Smith *et al.*, 2015). Some studies present the sizes of fish consumed by seals, e.g. to investigate the prey-size overlap between seals and fisheries (Hammond and Grellier, 2006; Hammond and Harris, 2006; Houle *et al.*, 2016; Lundström *et al.*, 2014; Lundström *et al.*, 2012).

3.3.2 Uncertainties in determining possible competition between fish predators and fisheries-lack of information on fish stocks

Diet data and individual prey consumption rates, together with data on population demography, can be used to produce estimates of total predation by the seal population, even leading to estimates of economic losses to the fisheries (Trijoulet *et al.*, 2018). However, estimates of actual prey availability, prey preferences of the predator population and the effect of the predators on the available prey stocks are often missing. In many cases there is little or no information on, e.g. size-specific catchability, migration, and behaviour of prey. Moreover, seasonal growth of the prey is often not taken into account although this could, at least partially, compensate the loss by predation (Aarts *et al*., 2018).

In many areas, assessments of fish stocks (i.e. seal prey) are based on fishery surveys carried out once a year on a generally coarse spatial scale (e.g. ICES rectangles), and the data are often approached as a static source. For example, these assessments do not account for growth of fish biomass throughout the year, which would potentially be available for predators and even less for the variability of this growth possibly affecting the whole system. Also, current data on fish stocks seldom include migration or any behavioural traits of the different prey species that might alter estimations of the effect of the predators. A good example is the sole (*Solea solea*) that migrates to shallower or deeper waters in the North Sea in response to temperature, becoming more or less available to seals and fisheries. Potentially, timing and scale of the prey movements will affect the prey availability to the predators and therefore the effect the predators might have on the total standing stocks. Hence, a better understanding of the ecological roles of different key ecological players, such as seals and fisheries, in the ecosystem is needed to better understand how they affect each other and to be able to apply a more ecosystem-based approach of fisheries management. Due to the complexity and dynamics of marine foodwebs, it should however be emphasised that ecological interactions are not simply about how much of species A is consumed/removed by species B.

In many areas, the current impact of seals on the marine ecosystem dynamics, in relation to anthropogenic effects, can be expected to be different from in the past, considering the growing seal population sizes and decreasing fisheries.

The extent to which consumption by seals impacts on the mortality of a prey population may also be affected by whether the seals “prefer” fish in poor condition, being easier to catch (less impact), or in good condition, being more energy rich (larger impact). Due to the lack of relevant studies, this topic is recommended for future research.

3.4 Ecological role/impact of indirect interactions

Following the collapses of fish stocks in the Northwest Atlantic and the increase in seal numbers, extensive consumption of fish by the seals has been suggested to have prevented the recovery of the fish stocks, despite long-term fishing bans (Bundy *et al.*, 2009). However, while some studies have presented support for seal impacts on fish stocks (O'Boyle and Sinclair, 2012; Chouinard *et al.*, 2005; Benoit *et al.*, 2011; Hammill *et al.*, 2014; Duplisea, 2005), others have not shown any strong relationships between seal predation and fish mortality (Mohn and Bowen, 1996; Trzcinski *et al.*, 2006; Fu *et al.*, 2001; Myers *et al.*, 1996; Buren *et al.*, 2014).

On the west coast of Scotland, the grey seal population has been suggested to be preventing the recovery of the cod stock, although reduced fishery catches would have been expected to permit stock recovery (Cook *et al.*, 2015; Cook and Trijoulet, 2016). Also, it was proposed that whiting and haddock stocks might also be affected by seal predation but to a smaller extent (Trijoulet *et al.*, 2017). However, a previous study, using another modelling approach, did not find support for seal predation impact on the same cod stock (Alexander *et al.*, 2015). In an assessment of resource competition between seals and fisheries in Irish waters, seal populations were considered to have minor impacts on fish stocks in general due to limited species and size overlap with fishery catches. Nevertheless, the seal populations were suggested to be potentially critical in the case of vulnerable fish stocks and for these species, increasing seal numbers might lead to more obvious competition with fisheries (Houle *et al.*, 2016).

In the North Sea, a recent paper investigates the impact of harbour seals on the local fish stocks in the Wadden Sea (Aarts *et al.*, 2018). Here, efforts were made to estimate the biomass available for predation, taking into account the efficiency of the fishing gear monitoring the fish stocks and correcting for the growth of fish between annual surveys. Based on the estimates provided, potential local effects are demonstrated. Even if the harbour seals spend only 13% of their diving time in the Wadden Sea, predation could cause an average annual mortality of 43% on fish in the Wadden Sea and 60% in the adjacent coastal zone, where they spend more of their foraging time.

In Iceland, there is a strong belief among different stakeholders that harbour seal predation has a large effect on salmonid angling. This is the main reason to cull harbour seals in Iceland. However, a report on bite marks from seals on salmonids (salmon, brown trout and charr) caught in five different rivers in NW Iceland, where seal predation has been suggested to be high, indicated that the proportion of seal injuries on caught salmonids was low (Granquist, 2014). Dietary studies using hard-part analysis (Granquist and Hauksson, 2016) and prey-DNA (Granquist, 2016; Granquist *et al.*, in prep.) similarly suggest that salmonids are not an important prey species for harbour seals in the area.

In the Baltic Sea, studies have shown that grey seal predation has low impact on offshore stocks of cod, herring and sprat compared with the impacts of environmental and anthropogenic factors (Eero *et al.*, 2015; MacKenzie *et al.*, 2011; Gårdmark *et al.*, 2012; Lindegren *et al.*, 2011; Östman *et al.*, 2014; Costalago *et al.*, in review). Estimates of ringed seal consumption of herring and vendace in the Bothnian Bay indicate that fish removal from seals might be comparable to, or even larger than, fishery catches (Lundström *et al.*, 2014). Furthermore, the choice of accounting for fish removal by seals or not doing so in the assessment model of Bothninan Bay vendace has impact on the assessment results and subsequent biological advice given to the management for the setting of fishery quotas for vendace in the area (Bergenius *et al.*, 2017, Lundström *et al.*, in prep.). In the Kattegat, preliminary estimates of cod consumption by harbour seals indicate that seals may have an important impact on the mortality and recruitment of this depleted cod stock (Lundström *et al.*, 2017).

Ecological modelling can be used as a means to study predator–prey dynamics, competition with fisheries and outcomes of different management strategies. Provided that adequate background information is available, e.g. about the diet, and realistic assumptions can be made e.g. on prey preference and available prey biomass, multi-species modelling can be a powerful tool to circumvent problems associated with large-scale manipulations of an ecosystem and contribute scientifically to the understanding of competition between seals and fisheries. For a thorough review of modelling approaches, see Plagányi (2007).

Different types of ecosystem models in which marine mammals have been included was reviewed by WGMME in 2015 (ICES, 2015), a few of which (e.g. Lassalle *et al.*, 2012) had been constructed specifically to examine marine mammal fishery interactions. However, the review did not cover the conclusions of these models in relation to the ecological role of marine mammals as top predators. For example, increased understanding of the role of marine mammals in regulating populations of lower trophic level species is needed, as well as their input into the detrital trophic web (e.g. through "whale fall"). An updated review of the ecological role of marine mammals, e.g. influence on structure, function and transfer of energy (and parasites) in marine foodwebs might be useful.

3.5 Seal impact on prey behaviour

Fishery actions have been suggested to change the behaviour and spatial distribution of fish stocks, and thus may affect the availability of prey to seals (Garrison and Link, 2000; Coetzee *et al.*, 2008). Seals are sometimes also perceived to have an indirect "scaring" effect on local fish abundance, which then would indirectly affect fisheries. It being difficult to collect such information, little evidence of this phenomenon exists in the literature. Benoit *et al.* (2010) studied the abundance pattern of polar cod in Franklin Bay and suggested that the behaviour of the fish depended on the presence of ringed seals in the area. They found that, during daytime, cod of all sizes aggregated in the deep inverse thermocline (160–230 m). From December (polar night) to April (18h daylight) smaller cod (<25 g) migrated into isothermal cold intermediate layer (90–150 m) during the night to avoid predation by immature seals (that are shallow diving). Large polar cod remained below 180 m at all times, which was suggested to be a way to minimize predation by mature deep-diving seals.

3.6 Transmission of nematode parasites

3.6.1 Introduction

Pinnipeds typically have four types of helminth parasites in their digestive tracts, namely acanthocephalans (thorny-headed worms), digeneans (flukes), nematodes (roundworms) and cestodes (tapeworms) (Vlasman and Campbell, 2004; Lehnert *et al*., 2007; 2010; 2016; Andersen-Ranberg *et al*., 2018; Waindok *et al*., 2018)). Here we focus on the nematodes. Nematode species found in the lungs, heart and other tissues of pinnipeds (e.g. Vlasman and Campbell, 2004) are beyond the scope of this short review.

Grey seals (*Halichoerus grypus*) are the main final host of the gastric parasitic nematodes cod worm or seal worm *Pseudoterranova decipiens* and the liver worm *Contracaecum osculatum* (Mehrdana *et al*., 2014). These nematode species are also found in bearded, common, harp, hooded and ringed seals (Bjorge, 1984; Valtonen *et al*., 1988; Marcogliese *et al*., 1996; Lehnert *et al*., 2007; Johansen *et al*., 2010; Brattey and Ni, 2011; Johansen, 2012). Genetic evidence suggests that three nematode species of the genus *Phocascaris*, reported from harp, hooded and ringed seals, should be considered as members of the genus *Contracaecum* (see Nadler *et al*., 2000; Abollo and Pascual, 2002; Mattiucci *et al*., 2008). The main other zoonotic nematode species in this region is the herring worm *Anisakis simplex*, for which cetaceans are the usual final hosts (Mattiucci *et al*., 2006; Mehrdana *et al*., 2014). Larvae of *Anisakis simplex* are however also recorded in harbour, grey, harp and ringed seals (Bjorge, 1984; Marcogliese *et al*., 1996; Johansen *et al*., 2010; Johansen, 2012; Zuo *et al*., 2018) and indeed Marcogliese *et al*. (1996) recorded a prevalence of 81% in a sample of 286 grey seal stomachs from the Gulf of St Lawrence. The congeneric *Anisakis pegreffii* is found in fish in the southern part of the ICES region, for example in hake from the Iberian Peninsula (Pascual *et al*., 2018) but in this area seals occur only as rare vagrants. The nematode *Trichinella*, commonly encountered in terrestrial systems, has been recorded in walrus as well as in ringed, bearded, and harp seals especially in the Arctic (Forbes, 2000).

Most marine mammals have low to moderate gastric nematode infections, which may be associated with ulcerative and granulomatous gastritis, eosinophilic, granulomatous and catarrhal-lymphocytic enteritis. High burdens of gastric nematodes tend to be associated with compromised immune systems and may be a consequence rather than a cause of poor health. In harbour porpoises in the Netherlands, there was a higher probability of parasite presence in the stomachs of porpoises in a poorer nutritive condition (ten Doeschate *et al*., 2017).

Harbour porpoises in the North and Baltic Seas suffer mostly from nematode and associated bacterial infections in the respiratory tract (Jepson *et al*., 2000, Jauniaux *et al*., 2002; Siebert *et al*., 2001; Wünschmann *et al*., 2001; Lehnert *et al*., 2005). Harbour porpoises in waters of Greenland, Iceland and Norway are less infected with pulmonary endoparasitosis and suffer generally less from impacts of parasitic associated lesions (Lehnert *et al*., 2005; Siebert *et al*., 2006; 2009; Barlow, per. comm). However, an increase in the severity of parasitic infections in harbour porpoises from Greenland was observed since a previous study in 1995, but still mostly associated with mild lesions (Lehnert *et al*., 2014). While *Anisakis simplex* was not found in 1995 it occurred in the 20 individuals sampled in 2009 (Lehnert *et al*., 2014). These authors argued that the increase of parasites was most likely associated with changes in diet, influenced by increasing sea temperatures and receding ice cover (Lehnert *et al*., 2014). Pathological and parasitological investigations of 385 stranded and bycaught harbour porpoises from the Baltic Sea revealed *Anisakis simplex*, *Contracaecum osculatum*, *Stenurus minor*, *Diphyllobothrium* sp., *Hysterothylacium aduncum* and *Pholeter gastrophilus* in the digestive tract (Siebert *et al*., in prep). However, infections of the respiratory tract and the derived lesions were identified as the best indicator for health condition of the harbour porpoise populations.

Des Clers and Wootten (1990) modelled the life cycle of *Pseudoterranova decipiens*. The model suggests that an increase in the number of seals would increase the number of adult parasites reproducing, hence the number of free-living larval stages. While the per capita transmission to the crustacean and fish levels would remain constant, relatively more parasites would reach the fish. The model necessarily makes simplifying assumptions and the fishery is the most important cause of changes in the number of fish in the system, but existence of a link between parasite abundance and seal abundance is probably inevitable. It should be noted that fisheries may exacerbate the problem of nematode infection in fish by discarding fish viscera loaded with live nematodes, which may then be ingested by other fish (and seabirds) (Gonzalez *et al*., 2018). In the German Wadden Sea, despite increased numbers of harbour and grey seals, a decrease of fish quality due to seal parasites has not been observed (Siebert, pers. Comm).

Fish are also involved in the transmission of metastrongyloid nematodes (lungworms) of harbour porpoises and harbour seals (Lehmann *et al*., 2010). As noted above, these infections do seem to have important implications for marine mammal health.

Of the ascaridoid nematodes found in seals and cetaceans, which have human health implications and potentially adverse consequences for fisheries, the impact of *Anisakis* is best documented. The larvae of these species infect marine fish, entering the musculature as well as the viscera, potentially to the detriment of both the fish and the human consumer. Ingestion of viable nematodes by humans presents health risks, of which the best known is anisakidosis or anisakiasis, the clinical symptoms of which may include acute pain, inflammation and vomiting; ingestion of *Pseudoterranova* has similar effects. Consumers may also become sensitised to *Anisakis* proteins (although no similar effect has been documented for *Pseudoterranova*), resulting in subsequent allergic reactions to anisakid proteins, which may include anaphylactic shock (see ESFA, 2010).

EU law requires that fish sold for human consumption are free of parasites; the necessity for inspection to eliminate heavily infested fish from the value chain, and removal of worms in fish processing factories, increases costs and is not always successful in preventing infected fish reaching the market. While freezing and cooking should kill the nematodes and at least partially denature the allergenic proteins, so the human health issues are largely controllable, the presence of worms in a fishmeal is also an aesthetic issue and there is concern in the fishing and associated food industry that greater awareness of the problem could lead to a fall in market demand for fish (see Karl, 2008).

3.6.2 Incidence in fish

Results of a recent large-scale survey of marine fish in European waters, several outputs from which are due to be published in the journal Fisheries Research, highlight the widespread occurrence of nematodes in species eaten by humans, especially by *Anisakis* spp., with some species such as hake routinely infected by large numbers of worms (Levsen *et al*., 2018; Pascual *et al*., 2018). Larger fish tend to have higher burdens of nematodes in their viscera and flesh and geographical patterns in infection rate are also apparent. Other nematode genera were also recorded, and reached high prevalence in certain species and areas, notably *Pseudoterranova* in cod from the North Sea and *Contracaecum* in haddock from the Barents Sea. However, the overwhelming dominance of *Anisakis* in many fish species examined suggests that, to the extent that the role of marine mammals is relevant, interactions between cetaceans and fisheries would be more important in this respect than interactions between seals and fisheries.

Recent studies from the Baltic, Iceland and Canada point to high abundances of *Pseudoterranova* and/or *Contracaecum* in cod as well as sometimes large numbers of *Anisakis simplex* (Marcogliese, 2001; Hauksson, 2011; Buchmann adn Kania, 2012; Mehrdana *et al*., 2014; Horbowy *et al*., 2016).

3.6.3 Fish condition and nematode infections

The ingestion of ascaridoid nematode larvae can have various effects on fish (see Buchmann and Mehrdana, 2016, for a review). In passing through the stomach wall and entering the viscera and musculature they can cause haemorrhaging and inflammation. The immune response may result in encapsulation of the larva. Large numbers within specific organs such as the liver can impair organ function. The infections may ultimately impair the physiological state, condition, health and survival of the fish. The swimming ability of the fish may be reduced, facilitating capture by marine mammals (to the advantage of the nematodes). *Pseudoterranova decipiens* produces pentanols and pentanones, having a potential anaesthetic effect on cod musculature, which may be connected with a reported decreased swimming performance. *Anisakis simplex* excretes molecules with an immunosuppressive effect, enabling its survival in the host (Bahlool *et al*., 2013; Zuo *et al*., 2018).

Horbowy *et al*. (2016) noted that the condition of infected cod was ~20% lower than that of uninfected fish, furthermore that prevalence and intensity of infection have dome-shaped relationships with fish length (i.e. with an apparent decline in the largest fish), which could be interpreted as suggesting the heavily infected fish mortality is higher. In relation to this observation, analysis for various fish species presented in Levsen *et al*. (2018) suggests that the relationships of infection prevalence and intensity with fish length may reach an asymptote but there was no clear indication of a decline in larger fish.

However, evidence of effects on fish condition is often equivocal. Mehrdana *et al*. (2014) recorded 100% prevalence of *Contracaecum osculatum* in cod from the Baltic Sea, with intensities in liver tissue of up to 320 worms per fish, but only a slight negative correlation between intensity and condition factor was noted.

3.6.4 The links between fish condition, mortality, parasites and seals

Several studies have highlighted an apparent link between high nematode burdens in fish and the proximity of seals colonies and/or increases in nematode burdens associated with increased seal abundance (e.g. Jensen and Idas, 1992; Hauksson, 2002; 2011; Buchmann and Kania, 2012; Mehrdana *et al*., 2014). However, at best these studies provide circumstantial evidence of a causal relationship. In the Wadden Sea, the region with highest seal populations in German waters, and where fish quality is routinely monitored, there have been no changes in fish quality (Siebert, pers. comm.).

In the Baltic, recent ecosystem developments and the regime shift in early 1990s, along with changes in cod life history have led to decreases in cod growth and condition as well as an increase of natural mortality. As a consequence, the analytical assessment of cod was considered to be unreliable (Eero *et al*., 2015). The main associated ecosystem changes were a decrease in the intensity of water exchange with North Sea, development of stagnation processes in the deep basins and changes in foodweb structure. In addition, the population of grey seals (the main predator of cod) has increased significantly since the beginning of the 2000s (Härkönen *et al*., 2013). Recent studies have revealed a significant increase in prevalence and intensity of parasites in livers of Baltic cod compared with the 1980s, when seal abundance was lower (Buchmann and Kania, 2012; Haarder *et al*., 2014; Nadolna and Podolska, 2014; Horbowy *et al*., 2016; Zuo *et al*., 2018).

However, when it comes to hard evidence linking seal abundance and nematode prevalence to fish condition, the picture becomes more complicated. Marcogliese (2001) argued that fluctuations in abundance of *Pseudoterranova decipiens* and other anisakids in fish and seals in the Gulf of St Lawrence (Canada) were probably related to climatic variation. Casini *et al*. (2016) analysed factors affecting cod condition in the Baltic during 1976–2014. These authors initially split the analysis into two time periods due to a shift in ecosystem structure and function in the early 1990s. The final model for 1976–1993 suggested that cod condition was determined by density-dependent processes, since there was a negative effect of high cod abundance. No density-dependent effect was apparent for 1994–2014, the final model included a positive effect of sprat biomass and a negative effect of the extent of hypoxic areas. In the model for the complete time-series, the effects of cod abundance and sprat abundance were weak, while a strong negative effect of seal abundance was apparent. Due to the somewhat contradictory nature of the results and the fact that the negative relationship between seal abundance and cod condition appeared to extend over only part of the time-series, the authors suggested that the effect of seal parasites on cod condition was probably minor at the population level compared with the other factors.

Evidently, statistically significant correlations do not prove causation and while a plausible mechanism exists to link seal abundance via levels of nematode infection in fish to effects on condition and mortality (see Figure 2), it is important to also consider other factors.

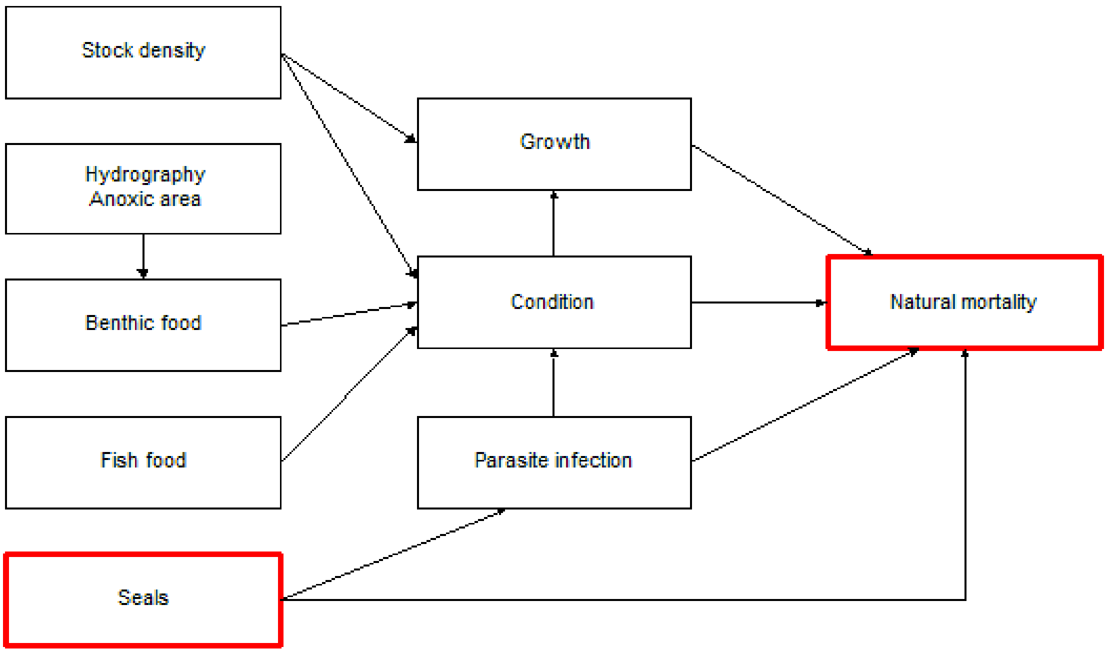

Figure 2. Simplified pathways of seal influence on condition and mortality of fish. Documented for Eastern Baltic cod. Diagram provided by Māris Plikšs.

3.6.5 Possible control measures

Buchmann and Mehrdana (2016) address the question of control measures that could be implemented to reduce nematode infections in fish stocks and thereby reduce the risk of human infections. They raise the question of "whether regulation of marine mammal populations in specified areas should be implemented in order to reduce the effect on fish stock size and fish product quality". As they observe, any such actions (e.g. culling, application of contraceptive measures) could conflict with the conservation of final host populations of species with protected status. Similarly, in the context of Baltic cod, Zuo *et al*. (2018) felt able to state that: "direct intervention, comprising regulation of the seal population by hunting, culling or targeted seal fishery, will be a solution, but may conflict with the protected status of grey seals".

Buchmann and Mehrdana (2016) suggest that mass treatment of final hosts (seals) with anthelmintics, so as to reduce the general infection pressure, might be an option, although it could raise environmental concerns as well as presenting logistical difficulties. However, the use of anthelmintics in farm animals, especially in small ruminants, but also increasingly in horses and cattle already presents important environmental and health issues, including the development of resistance in the parasites (Kaplan, 2004). For example, avermectins are excreted unchanged via faeces and the drug residues affect other trophic levels negatively. Ivermectin is documented to induce toxic effects on terrestrial and marine organisms, especially dung feeders and aquatic sediment invertebrates, potentially accumulating in benthic invertebrate fauna, due to its poor solubility in water (Sanderson *et al*., 2007; Michael *et al*., 2015). Furthermore, beneficial effects of treatment with anthelmintics last only for a specific time due to excretion and re-infection with nematodes with further prey intake.

Seals seem to adapt to infections with certain parasites such as *Pseudoterranova decipiens* and support more severe infections when aging, instead of developing an immunity to re-infection as observed for lung nematodes (McClelland, 2002; Lehnert *et al*., 2007; Ulrich *et al*., 2015). The animals seem to become desensitised to *Pseudoterranova decipiens* after shedding at least part of their gastric nematode burden during fasting periods like breeding or moulting (McClelland, 1980). Assuming that high seal abundance is an issue, a more benign solution would be the improvement of living conditions for marine mammals, as the intensity of parasitic infection corresponds to the immune competence of the host. This could be achieved by reductions in aquatic pollution (Stringer *et al*., 2014) or prevention of malnutrition (e.g. by avoiding overfishing) in order to strengthen the immune system.

3.6.6 Conclusion

There is a clear need for long-term simultaneous monitoring of nematode infection levels in fish and marine mammals to improve understanding of their interrelationship. However, as in most ecological studies of competition, a strong possibility remains that comparative observations (over time or comparing different areas) will not yield conclusive results. Model-based simulation studies, for example using a dynamic and quantitative implementation of the des Clers and Wootten (1990) life cycle model, ideally expanded into a full ecosystem model of parasite flow, could provide answers about the effects of changing seal abundance. Finally, however, studies of the effects of significant perturbation of the system, may be needed. Hypothetically, these could involve a substantial drop in fishing pressure, a seal cull or (in harbour seals) occurrence of a phocine distemper epizootic.

Seals routinely figure as a scapegoat when fisheries run into difficulties and this is not a modern phenomenon. Lofthouse (1887) describes damage by seals to salmon fishing in the River Tees in England and refers to 16th century accounts of seals being "very injurious to salmon". In general, both fishing and seal predation contribute to fish mortality. As Cook and Trijoulet (2016) observed in the case of the depleted cod population on the west coast of Scotland, "total mortality… is high enough to either cause population decline or prevent recovery. Reducing that total mortality can be influenced by human intervention, but how that intervention occurs will depend on the relative value of seals, cod, and the fishery to society". In relation to codworm, evidence that high seal abundance has a negative effect of fish mortality and/or condition via transmission of nematode parasites, is less secure than evidence of effects due to predation. Ultimately however, the choice of a solution also requires a value judgement.

3.7 Economic impact/financial losses to fisheries due to indirect interactions with seals

The perception that seals are in direct competition with fisheries is widespread in society and it is often suggested that seals have an economic impact on the fishing industry, although this is hard to investigate and quantify due to several factors. For example, it is often assumed that if seal numbers are reduced, more fish can be caught by fishermen. This assumes the seals to be the only predators and that indirect ecological interactions (e.g. increases in predators and competitors of the fish in question due to removal of seals) would not reduce fish availability. Furthermore, if fish abundance increased it is not clear that market prices for the fish would remain at their present level or indeed that increased fish abundance would be sustained, given the likelihood of increased fishing mortality due to increasing fishing pressure in response to increased fish abundance.

Trijoulet *et al.* (2018) investigated the economic effect of seal predation on demersal fisheries west of Scotland that traditionally targeted cod, haddock and whiting. The results show that large cod-fish trawlers were most sensitive to seal predation, while seal impacts were minor at the aggregate fishery level. The authors found that seal predation effects on revenues are small, although depending on the assumed foraging model of the seals (Type II functional response increased the sensitivity for seal predation for the fishery), which is an area where more research is needed.

Economic losses due to nematode parasites in fish are difficult to quantify, and quantifying the role of seals in this loss is even more problematic. There are real monetary costs to the fishing and fish processing industry due to the need to screen fish for nematodes and due to heavily parasitized fish being deemed unfit for human consumption. There are also costs in terms of human health and health care when nematodes slip through the control system and are consumed by humans. However, at present we have no clear basis on which to attribute any of these costs to seals. Even if it were concluded that high seal numbers result in economic losses due to higher parasite burdens in fish, appropriate remedial measures are currently unclear. Aside from legal protection afforded to seals in some areas (e.g. EU waters), as evident from the foregoing discussion, the economic and ecological consequences of culling seals (and the economic costs of the ecological consequences) are essentially unknown.

3.8 References

Aarts, G., Brasseur, S., Poos, J. J., Schop, J., Kirkwood, R., van Kooten, T., Mul, E., Reijnders, P., Rijnsdorp, A.D. and Tulp, I. 2018. Harbour seals are regaining top–down control in a coastal ecosystem. bioRxiv 267567; doi: https://doi.org/10.1101/267567.

Abollo, E. and Pascual, S. 2002. An SEM study of *Phocascaris cystophorae* Berland, 1964 (Nematoda: Anisakidae), a parasite of the hooded seal *Cystophora cristata*. Systematic Parasitology 51, 155-158.

Alexander, K. A., Heymans, J. J., Magill, S., Tomczak, M. T., Holmes, S. J., and Wilding, T. A. 2015. Investigating the recent decline in gadoid stocks in the west of Scotland shelf ecosystem using a foodweb model. Ices Journal of Marine Science, 72: 436–449.

Andersen-Ranberg E., Lehnert K., S. Leifsson P., Dietz R., Andersen S., Siebert U., Measures L., Sonne C. 2018. Morphometric, molecular and histopathologic description of hepatic infection by *Orthosplanchnus arcticus* (Trematoda: Digenea: Brachycladiidae) in ringed seals (*Pusa hispida*) from Northwest Greenland, Polar Biology, https://doi.org/10.1007/s00300-017-2245-6.

Andreasen, H., Ross, S. D., Siebert, U., Andersen, N. G., Ronnenberg, K. and Gilles, A. 2017. Diet composition and food consumption rate of harbor porpoises (*Phocoena phocoena*) in the western Baltic Sea. Marine Mammal Science, 33: 1053–1079.

Bahlool, Q.M., Skovgaard, A., Kania, P.W., Buchmann, K. 2013. Effects of excretory/secretory products from *Anisakis simplex* (Nematoda) on immune gene expression in rainbow trout (*Oncorhynchus mykiss*). Fish & Shellfish Immunology 35, 734–739.

Benoit, D., Simard, Y., Gagne, J., Geoffroy, M., and Fortier, L. 2010. From polar night to midnight sun: photoperiod, seal predation, and the diel vertical migrations of polar cod (*Boreogadus saida*) under landfast ice in the Arctic Ocean. Polar Biology, 33: 1505–1520.

Benoit, H. P., Swain, D. P., Bowen, W. D., Breed, G. A., Hammill, M. O., and Harvey, V. 2011. Evaluating the potential for grey seal predation to explain elevated natural mortality in three fish species in the southern Gulf of St Lawrence. Marine Ecology-Progress Series, 442: 149–167.

Bergenius, M., Lundström, K., and Cardinale, M. 2017. Biologisk rådgivning för siklöja i Bottenviken. Biologiskt råd till Havs- och vattenmyndigheten. In Swedish. 29 pp.

Bjorge, A. 1984. Parasitic nematodes in stomachs of grey seals, *Halichoerus grypus*, and common seals, *Phoca vitulina*, along the Norwegian coast. ICES CM 1984/N:3, 8 pp.

Brattey, J. and Ni, I-H. 2011. Ascaridoid nematodes from the stomach of harp seals, *Phoca groenlandica*, from Newfoundland and Labrador. Canadian Journal of Fisheries and Aquatic Sciences, 1992, 49(5): 956–966, https://doi.org/10.1139/f92-106 .

Buchmann, K., and Kania, P.W. 2012. Emerging *Pseudoterranova decipiens* (Krabbe, 1878) problems in Baltic cod, *Gadus morhua* L., associated with grey seal colonization of spawning grounds. Journal of Fish Diseases, 35: 861–866.

Bundy, A., Heymans, J. J., Morissette, L., and Savenkoff, C. 2009. Seals, cod and forage fish: A comparative exploration of variations in the theme of stock collapse and ecosystem change in four Northwest Atlantic ecosystems. Progress in Oceanography, 81: 188–206.

Buren, A. D., Koen-Alonso, M., and Stenson, G. B. 2014. The role of harp seals, fisheries and food availability in driving the dynamics of northern cod. Marine Ecology Progress Series, 511: 265–284.

Chouinard, G. A., Swain, D. P., Hammill, M. O., and Poirier, G. A. 2005. Covariation between grey seal (*Halichoerus grypus*) abundance and natural mortality of cod (*Gadus morhua*) in the southern Gulf of St Lawrence. Canadian Journal of Fisheries and Aquatic Sciences, 62: 1991–2000.

Coetzee, J. C., van der Lingen, C. D., Hutchings, L., and Fairweather, T. P. 2008. Has the fishery contributed to a major shift in the distribution of South African sardine? Ices Journal of Marine Science, 65: 1676–1688.

Coltman, D. W., Bowen, W. D., Boness, D. J., and Iverson, S. J. 1997. Balancing foraging and reproduction in the male harbour seal, an aquatically mating pinniped. Animal Behaviour, 54: 663–678.

Cook, R. M., Holmes, S. J., and Fryer, R. J. 2015. Grey seal predation impairs recovery of an over-exploited fish stock. Journal of Applied Ecology, 52: 969–979.

Cook, R. M., and Trijoulet, V. 2016. The effects of grey seal predation and commercial fishing on the recovery of a depleted cod stock. Canadian Journal of Fisheries and Aquatic Sciences, 73: 1319–1329.

Des Clers, S.A. and Wootten, R. 1990. Modelling the population dynamics of the sealworm *Pseudoterranova decipiens*. Netherlands Journal of Sea Research 25, 291–299.

Duplisea, D. E. 2005. Running the gauntlet: the predation environment of small fish in the northern Gulf of St Lawrence, Canada. Ices Journal of Marine Science, 62: 412–416.

Eero, M., Hjelm, J., Behrens, J., Buchmann, K., Cardinale, M., Casini, M., Gasyukov, P., *et al*. 2015. Eastern Baltic cod in distress: biological changes and challenges for stock assessment. Ices Journal of Marine Science, 72: 2180–2186.

ESFA. 2010. Scientific Opinion on risk assessment of parasites in fishery products. ESFA Journal 8, 1453, 91 pp.

Florin, A. B., Bergström, U., Ustups, D., Lundström, K., and Jonsson, P. R. 2013. Effects of a large northern European no-take zone on flatfish populations. Journal of Fish Biology, 83: 939–962.

Forbes, L.B. 2000. The occurrence and ecology of *Trichinella* in marine mammals. Veterinary parasitology, 93(3–4), 321–334.

Fu, C., Mohn, R., and Fanning, L. P. 2001. Why the Atlantic cod (*Gadus morhua*) stock off eastern Nova Scotia has not recovered. Can. J. Fish. Aquat. Sci., 58: 1613–1623.

Garrison, L. P., and Link, J. S. 2000. Fishing effects on spatial distribution and trophic guild structure of the fish community in the Georges Bank region. Ices Journal of Marine Science, 57: 723–730.

González, A.F., Gracia, J., Miniño, I., Romón, J., Larsson, C., Maroto, J., Reguiera, M. and Pascual, S. 2018. Approach to reduce the zoonotic parasite load in fish stocks: when science meets technology. Fish. Res. 202, 140–148.

Granquist, S. 2014a. Ummerki eftir sel á veiddum laxfiskum í völdum laxveiðiám í Húnaþingi vestra og Austur-Húnavatnssýslu [Injuries from seal on caught salmonids in selected rivers in Húnaþingi vestra og Austur-Húnavatnssýslu, Iceland]. Institute of Freshwater Fisheries, VMST/14019.

Granquist, S. M. 2016. Ecology, tourism and management of harbour seals (*Phoca vitulina*). PhD thesis Stockholm University.

Granquist, S. M., and Hauksson, E. 2016. Diet of harbour seals in a salmon estuary in North-West Iceland. Icelandic Agricultural Sciences, 29: 7–19.

Gårdmark, A., Östman, Ö., Nielsen, A., Lundström, K., Karlsson, O., Ponni, J., and Aho, T. 2012. Does predation by grey seals (*Halichoerus grypus*) affect Bothnian Sea herring stock estimates? Ices Journal of Marine Science, 69: 1448–1456.

Haarder, S., Kania, P. W., Galatius, A., and Buchmann, K. 2014. Increased *Contracaecum osculatum* infection in Baltic cod (*Gadus morhua*) livers (1982–2012) associated with increasing grey seal (*Halichoerus grypus*) populations. Journal of Wildlife Diseases, 50: 537–543.

Hammill, M. O., and Stenson, G. B. 2000. Estimated prey consumption by harp seals (*Phoca groenlandica*), hooded seals (*Cystophora cristata*), grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) in Atlantic Canada. J. Northw. Atl. Fish. Sci., 26: 1–23.

Hammill, M. O., Stenson, G. B., Swain, D. P., and Benoit, H. P. 2014. Feeding by grey seals on endangered stocks of Atlantic cod and white hake. Ices Journal of Marine Science, 71: 1332–1341.

Hammond, P. S., and Grellier, K. 2006. Grey seal diet composition and prey consumption in the North Sea. Final report to Department for Environment Food and Rural Affairs on project MF0319. 54 pp.

Hammond, P. S., and Harris, R. N. 2006. Grey seal diet composition and prey consumption off western Scotland and Shetland. Final report to Scottish Executive Environment and Rural Affairs Department and Scottish Natural Heritage: 41 pp.

Hansson, S., Bergström, U., Bonsdorff, E., Härkönen, T., Jepsen, N., Kautsky, L., Lundström, K., *et al*. 2017. Competition for the fish–fish extraction from the Baltic Sea by humans, aquatic mammals, and birds. Ices Journal of Marine Science: fsx207-fsx207.

Härkönen T, Galatius A, Bräeger S, Karlsson O and Ahola M. 2013. Population growth rate, abundance and distribution of marine mammals. HELCOM Core Indicator Report. URL http://www.helcom.fi/Core%20Indicators/HELCOM-CoreIndicator-Population_growth_rate_abundance_and_distribution_of_marine_mammals.pdf.

Härkönen, T., and Heide-Jørgensen, M.-P. 1991. The harbour seal *Phoca vitulina* as a predator in the Skagerrak. Ophelia, 34: 191–207.

Hauksson, E. 2002. Decreases in sealworm (*Pseudoterranova* sp.) abundance in short-spined sea scorpion (*Myoxocephalus scorpius*) following declines in numbers of seals at Hvalseyjar, western Iceland. Polar Biol. 25, 531–537.

Hauksson, E. 2011. The prevalence, abundance, and density of *Pseudoterranova* sp. (p) larvae in the flesh of cod (*Gadus morhua*) relative to proximity of grey seal (*Halichoerus grypus*) colonies on the coast off Drangar, Northwest Iceland. Journal of Marine Biology (2011), Article ID 235832, 8 pp.

Hedd, A., Gales, R., and Renouf, D. 1997. Inter-annual consistency in the fluctuating energy requirements of captive harp seals *Phoca groenlandica*. Polar Biology, 18: 311–318.

Horbowy, J., Podolska, M. and Nadolna-Ałtyn, K. 2016. Increasing occurrence of anisakid nematodes in the liver of cod (*Gadus morhua*) from the Baltic Sea: Does infection affect the condition and mortality of fish? Fisheries Research 179, 98–103.

Houle, J. E., de Castro, F., Cronin, M. A., Farnsworth, K. D., Gosch, M., and Reid, D. G. 2016. Effects of seal predation on a modelled marine fish community and consequences for a commercial fishery. Journal of Applied Ecology, 53: 54–63.

ICES. 2006. Report of the Working Group on Marine Mammal Ecology (WGMME), 30 January–2 February 2006, ICES Headquarters. ACE:06. 55 pp.

ICES. 2015. Report of the Working Group on Marine Mammal Ecology (WGMME), 9–12 February 2015, London, UK. ICES CM 2015/ACOM:25. 114 pp.

ICES. 2017a. Report of the Working Group on Marine Mammal Ecology (WGMME), 6–9 February 2017, St Andrews, Scotland, UK. ICES CM 2017/ACOM:27. 102 pp.

ICES. 2017b. Report of the Working Group on Predator–prey Interactions between Grey Seals and other marine mammals (WKPIGS), Middelfart, DK. ICES CM 2017/SSGEPD:18. 26 pp.

Innes, S., Lavigne, D. M., Earle, W. M., and Kovacs, K. M. 1987. Feeding rates of seals and whales. Journal of Animal Ecology, 56: 115–130.

Jauniaux, T., Petitjean, D., Brenez, C., Borrens, M., Brosens, L., Haelters, J., Tavernier, T. and Coignoul, F. 2002. Post-mortem findings and causes of death of harbour porpoises (*Phocoena phocoena*) stranded from 1990 to 2000 along the coastlines of Belgium and Northern France. *Journal of Comparative Pathology*, **126**, 243–253.

Jeanniard-du-Dot, T., Trites, A. W., Arnould, J. P. Y., Speakman, J. R., and Guinet, C. 2017. Activity-specific metabolic rates for diving, transiting, and resting at sea can be estimated from time–activity budgets in free-ranging marine mammals. Ecology and Evolution, 7: 2969–2976.

Jensen, T. and Idås, K., 1992. Infection with *Pseudoterranova decipiens* (Krabbe, 1878) larvae in cod (*Gadus morhua*) relative to proximity of seal colonies. Sarsia 76, 227–230.

Jepson, P. D., Baker, J. R., Kuiken, T., Simpson, V. R., Kennedy, S. and Bennett, P. M. 2000. Pulmonary pathology of harbour porpoises stranded in England and Wales between 1990 and 1996. *Veterinary Record*, **146**, 721–728.

Johansen, C.E., Lydersen, C., Aspholm, P.E., Haug, T. and Kovaks, K.M. 2010. Helminth parasites in ringed seals (*Pusa hispida*) from Svalbard, Norway with special emphasis on nematodes: variation with age, sex, diet, and location of host. Journal of Parasitology 96, 946–953.

Johansen, J.D. 2012. Nematode infection in harbour seal (*Phoca vitulina*) at two sites; Sandøy and Hvaler archipelago, Norway. Master thesis, University of Oslo, 69 pp.

Kaplan, R. M. 2004. Drug resistance in nematodes of veterinary importance: a status report. Trends in parasitology, 20(10), 477–481.

Karl, H. 2008. Nematode larvae in fish on the German market: 20 years of consumer-related research. Archiv für Lebensmittelhygiene 59, 107–116.

Kaschner, K., Watson, R., Christensen, V., Trites, A. W., and Pauly, D. 2001. Modeling and mapping trophic overlap between marine mammals and commercial fisheries in the North Atlantic. In D. Zeller and R. D. Pauly Watson (Eds), Fisheries impacts on North Atlantic ecosystems: catch, effort and national/regional datasets. Fisheries Centre Research, 9: 35–45.

Kastelein, R. A., Staal, C., and Wiepkema, P. R. 2005. Food consumption and body mass of captive harbor seals (*Phoca vitulina*). Aquatic Mammals, 31: 34–42.

Kauppinen, T., Siira, A., and Suuronen, P. 2005. Temporal and regional patterns in seal-induced catch and gear damage in the coastal trap-net fishery in the northern Baltic Sea: effect of netting material on damage. Fisheries Research, 73: 99–109.

Lassalle, G., Gascuel, D., Le Loc'h, F., Lobry, J., Pierce, G. J., Ridoux, V., Santos, M. B., *et al*. 2012. An ecosystem approach for the assessment of fisheries impacts on marine top predators: the Bay of Biscay case study. Ices Journal of Marine Science, 69: 925–938.

Lehnert K., Raga J.A., Siebert U. 2005. Macroparasites in stranded and bycaught harbour porpoises from German and Norwegian waters. *Diseases of Aquatic Organisms* 64, 265–269.

Lehnert K., Raga J.A., Siebert U. 2007. Parasites in harbour seals (*Phoca vitulina*) from the German Wadden Sea between two phocine distemper epidemics. *Helgoland Marine Research* DOI: 10.1007/s10152-007-0072-9.

Lehnert K., Fonfara S., Wohlsein P., Siebert U. 2007. *Isocyamus delphinii*: First record of whale-lice on a harbour porpoise (*Phocoena phocoena*) from German waters. *Veterinary Record* 161, 526–528.

Lehnert K., Schwanke E., Hahn K., Wohlsein P., Siebert U. 2016. Heartworm (*Acanthocheilonema spirocauda*) and seal louse (*Echinophthirius horridus*) infections in harbour seals (*Phoca vitulina*) from the North and Baltic Sea, *Journal of Sea Research* 113, 65–72.

Lehnert K., Seibel H., Hasselmeier I., Wohlsein P., Iversen M., Nielsen N.H., Heide-Jørgensen M.P., Prenger-Berninghoff E., Siebert U. 2014. Increase in parasite burden and associated pathology in harbour porpoises (*Phocoena phocoena*) in West Greenland. *Polar Biology* 37, 321–331.

Lehnert K., van Samson-Himmelstjerna G., Schaudien D., Bleidorn C., Wohlsein P., Siebert U. 2010. Transmission of lungworms of harbour porpoises and harbour seals: Molecular tools determine potential vertebrate intermediate hosts. *International Journal for Parasitology* 40, 845–853.

Levsen, A., Svanevik, C.S., Cipriani, P., Mattiucci, S., Gay, M., Hastie, L.C., Bušelić, I. Mladineo, I., Karl, H., Ostermeyer, U. Buchmann, K., Højgaard, D.P., González, A.F., Pascual, S. and Pierce, G.J. 2018. A survey of zoonotic nematodes of commercial key fish species from major European fishing grounds - introducing the FP7 PARASITE exposure assessment study. Fisheries Research.

Lindegren, M., Östman, Ö., and Gårdmark, A. 2011. Interacting trophic forcing and the population dynamics of herring. Ecology, 92: 1407–1413.

Lofthouse, R. 1887. The river Tees: its marshes and their fauna. Naturalist, Hull (1887) 1–16.

Lundström, K., Bergenius, M., Aho, T., and Lunneryd, S. G. 2014. Födoval hos vikaresäl i Bottenviken: Rapport från den svenska forskningsjakten 2007–2009. Sveriges lantbruksuniversitet, Lysekil. Aqua reports, 2014: 23 pp.

Lundström, K., Hjerne, O., and Karlsson, O. 2012. Grey seal (*Halichoerus grypus*) prey consumption in the Baltic Sea. In: Assessment of dietary patterns and prey consumption of marine mammals: Grey seals (*Halichoerus grypus*) in the Baltic Sea. PhD. thesis, Department of Biological and Environmental Sciences, University of Gothenburg. 30 pp.

Lundström, K., Lövgren, J., and Eero, M. 2017. Kattegat cod numbers eaten by harbour seals. In ICES. 2017. Report of the Benchmark Workshop on Baltic Stocks (WKBALT), 7–10 February 2017, Copenhagen, Denmark. ICES CM 2017/ACOM:30. 108 pp: 97–107.

Lydersen, C., Hammill, M. O., and Kovacs, K. M. 1995. Milk Intake, Growth and Energy-Consumption in Pups of Ice-Breeding Grey Seals (*Halichoerus grypus*) from the Gulf of St Lawrence, Canada. Journal of Comparative Physiology B-Biochemical Systemic and Environmental Physiology, 164: 585–592.

MacKenzie, B. R., Eero, M., and Ojaveer, H. 2011. Could Seals Prevent Cod Recovery in the Baltic Sea? PLoS ONE, 6.

Marcogliese, D.J. 2001. Distribution and abundance of sealworm (*Pseudoterranova decipiens*) and other anisakid nematodes in fish and seals in the Gulf of St Lawrence: potential importance of climatic conditions. *NAMMCO Sci. Publ.* 3: 113–128.

Marcogliese, D.J., Boily, F. and Hammill, M.O. 1996. Distribution and abundance of stomach nematodes (Anisakidae) among grey seals (*Halichoerus grypus*) and harp seals (*Phoca groenlandica*) in the Gulf of St Lawrence. Can. J. Fish. Aquat. Sci. 53: 2829–2836.

Mattiucci, S. and Nascetti, G. 2006. Molecular systematics, phylogeny and ecology of anisakid nematodes of the genus *Anisakis* Dujardin, 1845: an update. Parasite, 13(2), 99–113.

Mattiucci, S., Paoletti, M., Webb, S.C., Sardella, N., Timi, J.T., Berland, B., Nascetti, G. 2008. Genetic relationships among species of *Contracaecum* Railliet and Henry, 1912 and *Phocascaris* Host, 1932 (Nematoda: Anisakidae) from pinnipeds based on mitochondrial Cox2 sequences, and congruence with allozyme data. Parasite 15, 408–419.

McClelland, G. 1980. *Phocanema decipiens*: growth, reproduction, and survival in seals. Experimental parasitology, 49(2), 175–187.

McClelland, G. 2002. The trouble with sealworms (*Pseudoterranova decipiens* species complex, Nematoda): a review. Parasitology, 124(7), 183–203.

Mehrdana, F., Bahlool, Q. Z., Skov, J., Marana, M. H., Sindberg, D., Mundeling, M., Overgaard, B.C. *et al*. 2014. Occurrence of zoonotic nematodes *Pseudoterranova decipiens*, *Contracaecum osculatum* and *Anisakis simplex* in cod (*Gadus morhua*) from the Baltic Sea. Veterinary Parasitology, 205: 581–587.

Michael, S. A., Chilvers, B. L., Roe, W. D., Gartrell, B. D. 2015. Long-term survival and reproductive success of New Zealand sea lions (*Phocarctos hookeri*) treated with ivermectin as pups. Wildlife Research, 42(8), 660–667.

Mohn, R., and Bowen, W. D. 1996. Grey seal predation on the eastern Scotian Shelf: modelling the impact on Atlantic cod. Can. J. Fish. Aquat. Sci., 53: 2722–2738.

Moore, J.E., Wallace, B.P., Lewison, R.L., Žydelis, R., Cox, T.M. and Crowder, L.B. 2009. A review of marine mammal, sea turtle and sea bird bycatch in USA fisheries and the role of policy in shaping management. Marine Policy 33, 435–451.

Myers, R. A., Hutchings, J. A., and Barrowman, N. J. 1996. Hypotheses for the decline of cod in the North Atlantic. Marine Ecology-Progress Series, 138: 293–308.

Nadler S.A., D'Amelio S., Fagerholm H.P., Berland B. and Paggi L. 2000. Phylogenetic relationships among species of *Contracaecum* Railliet & Henry, 1912 and *Phocascaris* Host, 1932 (Nematoda: Ascaridoidea) based on nuclear rDNA sequence data. Parasitology, 2000, 121, 455–463.

Nadolna, K., and Podolska, M. 2014. Anisakid larvae in the liver of cod (*Gadus morhua* L.) from the Southern Baltic Sea. Journal of Helminthology, 88: 237–246.

Näreaho, A., Eriksson-Kallio, A. M., Heikkinen, P., Snellman, A., Sukura, A., Koski, P. 2017. High prevalence of zoonotic trematodes in roach (*Rutilus rutilus*) in the Gulf of Finland. Acta Veterinaria Scandinavica, 59(1), 75.

Neimanis, A. S., Moraeus, C., Bergman, A., Bignert, A., Höglund, J., Lundström, K., Strömberg, A., Bäcklin, B. M. 2016. Emergence of the zoonotic biliary trematode *Pseudamphistomum truncatum* in grey seals (*Halichoerus grypus*) in the Baltic Sea. PloS one, 11(10), e0164782.

Northridge, S.P. 1984. World review of interactions between marine mammals and fisheries. FAO Fisheries Technical Paper. No. 251. Rome, FAO, 190 pp.

Northridge, S.P. 1991. An updated world review of interactions between marine mammals and fisheries. FAO Fisheries Technical Paper. No. 251, Suppl. 1. Rome, FAO. 58 pp.

O'Boyle, R., and Sinclair, M. 2012. Seal-cod interactions on the Eastern Scotian Shelf: Reconsideration of modelling assumptions. Fisheries Research, 115: 1–13.

Östman, Ö., Karlsson, O., Pönni, J., Kaljuste, O., Aho, T., and Gårdmark, A. 2014. Relative contributions of evolutionary and ecological dynamics to body size and life-history changes of herring (*Clupea harengus*) in the Bothnian Sea. Evolutionary Ecology Research, 16: 417–433.

Overholtz, W. J., and Link, J. S. 2007. Consumption impacts by marine mammals, fish, and seabirds on the Gulf of Maine-Georges Bank Atlantic herring (*Clupea harengus*) complex during the years 1977–2002. Ices Journal of Marine Science, 64: 83–96.

Pascual, S., Rodríguez, H., Pierce, G.J., Hastie, L.C. & González, A.F. 2018. The NE Atlantic hake: a neglected high exposure risk for zoonotic parasites in European fish markets. Fisheries Research 202, 69–78.

Pittman, A., Galatius, A., Teilmann, J., Dietz, R., Jarnit, S., Andersen, S.M., Murray, D., Zepeda, L., Fordyce, S.L., Haile, J., Møller, P.R., Bunce, M., Thomas, M., Gilbert, P. and Olsen, M.T. When protected becomes pest: Molecular diet analysis of grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) in Denmark in light of the seal-fisheries conflict.

Plagányi, É. E. 2007. Models for an ecosystem approach to fisheries. FAO Fisheries Technical Paper. No. 477. Rome, FAO. 108 pp.

Punt, A. E., and Butterworth, D. S. 1995. The effects of future consumption by the Cape fur seal on catches and catch rates of the cape hakes. 4. Modelling the biological interaction between Cape fur seals *Arctocephalus pusillus pusillus* and the cape hakes *Merluccius capensis* and M-paradoxus. South African Journal of Marine Science-Suid-Afrikaanse Tydskrif Vir Seewetenskap, 16: 255–285.

Read, A.J. 2008. The looming crisis: interactions between Marine mammals and fisheries. Journal of Mammalogy 89, 541–548.

Reilly, J. J., Fedak, M. A., Thomas, D. H., Coward, W. A. A., and Anderson, S. S. 1996. Water balance and the energetics of lactation in grey seals (*Halichoerus grypus*) as studied by isotopically labelled water methods. Journal of Zoology (London), 238: 157–165.

Ruzicka, J. J., Steele, J. H., Ballerini, T., Gaichas, S. K., and Ainley, D. G. 2013. Dividing up the pie: Whales, fish, and humans as competitors. Progress in Oceanography, 116: 207–219.

Ryg, M., and Øritsland, N. A. 1991. Estimates of energy expenditure and energy consumption of ringed seals (*Phoca hispida*) throughout the year. Polar Research, 10: 595–601.

Sanderson, H., Laird, B., Pope, L., Brain, R., Wilson, C., Johnson, D., Bryning, G., Peregrine, A. S., Boxall, A., Solomon, K. 2007. Assessment of the environmental fate and effects of ivermectin in aquatic mesocosms. Aquatic toxicology, 85(4), 229–240.

Santos, M.B., Saavedra, C., and Pierce, G.J. 2014. Quantifying the predation on sardine and hake by cetaceans in the Atlantic waters of the Iberian Peninsula. Deep Sea Research Part II: Topical Studies in Oceanography, 106: 232–244.

Scharff-Olsen, C.J., Andersen, S.M., Galatius, A., Teilmann, J., Dietz, R., Jarnit, S., Pittman, A., Møller, P.R., Lundström, K., Olsen, M.T. The diet of seals in the Baltic Sea region: A review of published and new data 1968–2013.

Siebert U., Wünschmann A., Weiss R., Frank H., Benke H., Frese K. 2001. Post-mortem findings in harbour porpoises (*Phocoena phocoena*) from the German North and Baltic Sea. *Journal of Comparative Pathology* 124, 102–114

Siebert U., Wünschmann A., Tolley K., Vikingsson G., Olafsdottir D., Lehnert K., Weiss R., Baumgärtner W. 2006. Pathological findings in harbour porpoises (*Phocoena phocoena*) originating from Norwegian and Icelandic waters. *Journal of Comparative Pathology* 134 (2-3), 134–142.

Siebert U., Wohlsein P., Lehnert K., Baumgärtner W. 2007. Pathological findings in Harbour Seals (*Phoca vitulina):* 1996–2005. *Journal of Comparative Pathology* 137, 47–58.

Siebert U., Prenger-Berrninghof E., Weiss R. 2009. Regional differences in bacteria flora in harbour porpoises from the North Atlantic: environmental effects. *Journal of Applied Microbiology* 106, 329–337.

Smith, L. A., Link, J. S., Cadrin, S. X., and Palka, D. L. 2015. Consumption by marine mammals on the Northeast US continental shelf. Ecological Applications, 25: 373–389.

Smout, S., Rindorf, A., Hammond, P. S., Harwood, J., and Matthiopoulos, J. 2014. Modelling prey consumption and switching by UK grey seals. Ices Journal of Marine Science, 71: 81–89.

Sparling, C. E., and Fedak, M. A. 2004. Metabolic rates of captive grey seals during voluntary diving. J. Exp. Biol., 207: 1615–1624.

Sparling, C. E., and Smout, S. C. 2003. Population energy budget for UK North Sea grey seals. SCOS Briefing Paper, 03/09: 58–62.

Sparling, C. E., Thompson, D., Fedak, M. A., Gallon, S. L., and Speakman, J. R. 2008. Estimating field metabolic rates of pinnipeds: doubly labelled water gets the seal of approval. Functional Ecology, 22: 245–254.

Stringer, A. P., Linklater, W. 2014. Everything in moderation: principles of parasite control for wildlife conservation. BioScience, 64(10), 932–937.

ten Doeschate MTI, Ijsseldijk, LL, Hiemstra, S, de Jong, EA, Strijkstra, A, Gröne A and Begeman L. 2017. Quantifying parasite presence in relation to biological parameters of harbour porpoises *Phocoena phocoena* stranded on the Dutch coast. Dis Aquat Organ. 127: 49–56.

Trijoulet, V., Dobby, H., Holmes, S. J., and Cook, R. M. 2018. Bioeconomic modelling of grey seal predation impacts on the West of Scotland demersal fisheries. Ices Journal of Marine Science: fsx235-fsx235.

Trijoulet, V., Holmes, S. J., and Cook, R. M. 2017. Grey seal predation mortality on three depleted stocks in the West of Scotland: What are the implications for stock assessments? Canadian Journal of Fisheries and Aquatic Sciences: 1–10.

Trites, A. W., Christensen, V., and Pauly, D. 1997. Competition between fisheries and marine mammals for prey and primary production in the Pacific Ocean. Journal of Northwest Atlantic Fishery Science, 22: 173–187.

Trzcinski, M. K., Mohn, R., and Bowen, W. D. 2006. Continued decline of an Atlantic cod population: How important is gray seal predation? Ecological Applications, 16: 2276–2292.

Ulrich, S. A., Janecek, E., Lehnert, K., Siebert, U. and Strube, C. 2015. A recombinant antigen-based enzyme-linked immunosorbent assay (ELISA) for lungworm detection in seals. *Parasites & Vectors* 8, 443.

Unger, B., Herr, H., Benke, H., Böhmert, M., Burkhardt-Holm, P., Dähne, M., Hillmann, M., Wolff-Schmidt, K., Wohlsein, P. and Siebert, U. 2017. Marine debris in harbour porpoises and seals from German waters. Marine Environmental Research 130, 77–84.

Valtonen, E.T., Fagerholm, H.-P. and Helle, E., *et al*. 1988. *Contracaecum osculatum* (Nematoda: Anisakidae) in fish and seals in Bothnian Bay (Northeastern Baltic Sea). International Journal for Parasitology 18, 365–370.

Vincent, C., Ridoux, V., Fedak, M. A., McConnell, B. J., Sparling, C. E., Leaute, J.-P., Jouma'a, J., *et al*. 2016. Foraging behaviour and prey consumption by grey seals (*Halichoerus grypus*)—spatial and trophic overlaps with fisheries in a marine protected area. Ices Journal of Marine Science, 73: 2653–2665.

Vlasman, K.L. and Campbell, G.D. 2004. Field Guide: Diseases and Parasites of Marine Mammals of the Eastern Arctic. Canadian Cooperative Wildlife Health Centre: Newsletters & Publications. 22. http://digitalcommons.unl.edu/icwdmccwhcnews/22 .

Waindok P., Lehnert K., Siebert U., Pawliczkac I., Strube C. 2018. Prevalence and molecular characterisation of *Acanthocephala* in pinnipedia of the North and Baltic Seas, Elsevier IJP Parasites and Wildlife, 7, 34–43.

Westerberg, H., Lunneryd, S. G., Fjälling, A., and Wahlberg, M. 2006. Reconciling fisheries activities with the conservation of seals throughout the development of new fishing gear: A case study from the Baltic fishery-grey seal conflict. American Fisheries Society Symposium, 2006: 587–597.

Wünschmann A., Siebert U., Frese K., Weiss R., Lockyer C., Heide-Jørgensen M.P., Weiss R., Müller G., Baumgärtner W. 2001. Evidence of infectious diseases in harbour porpoises (*Phocoena phocoena*) hunted in the waters of Greenland and by-caught in the German North and Baltic Sea. *Veterinary Record* 148, 715–720.

Zuo, S., Kania, P. W., Mehrdana, F., Marana, M. H., Buchmann, K. 2018. *Contracaecum osculatum* and other anisakid nematodes in grey seals and cod in the Baltic Sea: molecular and ecological links. Journal of helminthology, 92(1), 81–89.

ToR B Appendix 1: Studies of seal diet in the ICES region 2006–2017

This section compiles information on the diet of seals in the ICES region. A similar assessment was carried out by WGMME in 2006 (ICES, 2006). The background to the review in the 2006 report was an assignment from the Study Group on Multispecies Assessments North Sea (SGMSNS) about prey consumption datasets for North Sea marine mammals to be included in multispecies assessments of the Working Group on the Multispecies Model of the North Sea (WGMSNS). As the report from 2006 contains publications until 2005, the current review is based on publications (published articles, reports, theses and recent studies) from 2006 and onwards. Studies looking at habitat selection and movement ecology are not included. Given the limited resources available to produce this review, some studies might have been overlooked.

Table A1. Seal diet studies in the ICES region reported between 2006 and 2017. A compilation of earlier seal diet studies was presented in the WGMME in 2006 (ICES, 2006).

Seal species	Area	Time	Comment	Reference
H. grypus	North Sea	1985, 2002		(Hammond and Grellier, 2006)
H. grypus	W Scotland	1985, 2002		(Hammond and Harris, 2006; Harris, 2007)
H. grypus	NW Atlantic	1993–2000		(Beck *et al.*, 2007)
H. grypus	NW Atlantic	1991–1998		(Bowen and Harrison, 2007)
H. grypus	NW Atlantic	1985–2004		(Hammill *et al.*, 2007)
H. grypus	Bothnian Sea	2007	In Swedish with English abstract	(Lagström, 2007)
H. grypus	Baltic Sea	2001–2005		(Lundström *et al.*, 2007; Lundström *et al.*, 2010)
H. grypus	Brittany, France	1998–2000		(Ridoux *et al.*, 2007)
H. grypus	NW Atlantic	1996–2001		(Tucker *et al.*, 2007; Tucker *et al.*, 2008)
H. grypus	Central Baltic Sea	2010	In Swedish with English abstract	(Asp, 2011)
H. grypus	NW Atlantic	1994–2008		(Hammill, 2011)
H. grypus	Baltic Sea	2001–2007	In Finnish with English summary and legends	(Kauhala *et al.*, 2011)
H. grypus	S Ireland	2009–2013		(Gosch *et al.*, 2014; Gosch, 2017)
H. grypus	NW Atlantic	1996–2011		(Hammill *et al.*, 2014)
H. grypus	French, Belgian and Dutch coastline	2003–2013	Predation on harbour porpoises based on stranding events	(Jauniaux *et al.*, 2014; Leopold *et al.*, 2015)
H. grypus	Brittany, France	2004–2011	Stranded animals	(Méheust *et al.*, 2015)

Seal species	Area	Time	Comment	Reference
H. grypus	Helgoland	2013–2014	Observations of grey seal preying upon harbour seals.	(van Neer *et al.*, 2015)
H. grypus	Isle of May	2014–2015		(Brownlow *et al.*, 2016)
H. grypus	Scotland, England	2010–2011		(Hammond and Wilson, 2016)
H. grypus	Scotland, England	1972–2008		(Hanson *et al.*, 2017)
H. grypus	N Atlantic		Review of current knowledge of grey seal predation on seals and harbour porpoises.	(ICES, 2017)
H. grypus	S Baltic Sea	2014–2016		(Zrust, 2017)
H. grypus	Baltic Sea	2011–2012	Subitted to PLOS ONE	Tverin *et al.*, submitted
P. vitulina	NE Scotland	2000		(Middlemas *et al.*, 2006)
P. vitulina	Limfjord; W Baltic Sea	1997–1998; 2001–2005		(Andersen *et al.*, 2007)
P. vitulina	SE Scotland	1998–2003		(Sharples *et al.*, 2009)
P. vitulina	W Ireland	2006–2007		(Kavanagh *et al.*, 2010)
P. vitulina	Normandy, France	2000–2004		(Spitz *et al.*, 2010)
P. vitulina	Scotland, England	2010–2012		(Wilson, 2014)
P. vitulina	English Channel, France	2002–2011		(Spitz *et al.*, 2015)
P. vitulina	Wadden Sea	2012–2014	Stranded animals	(de la Vega *et al.*, 2016; de la Vega *et al.*, 2018)
P. vitulina	NW Iceland	2009–2011		(Granquist and Hauksson, 2016)
P. vitulina	Scotland, England	2010–2011		(Wilson and Hammond, 2016b)
P. vitulina	S Norway	2015–2016		(Sørlie, 2017)
P. vitulina	Wadden Sea	2002–2009		(Aarts *et al.*, 2018)
P. groenlandicus	NW Atlantic			(Marshall *et al.*, 2010)
P. groenlandicus	Svalbard	1996–2006		(Lindstrøm *et al.*, 2012)
P. groenlandicus				(Haug *et al.*, 2017)
C. cristata	E Greenland	1987; 1999–2003		(Haug *et al.*, 2007)
P. hispida	Bothnian Bay			(Sinisalo *et al.*, 2006; Sinisalo, 2007; Sinisalo *et al.*, 2008)
P. hispida	Svalbard	2002–2004		(Labansen *et al.*, 2007)
P. hispida	E Greenland	2002–2004		(Labansen *et al.*, 2011)
P. hispida	Hudson Bay	2009–2011		(Young and Ferguson, 2013)

Seal species	Area	Time	Comment	Reference
P. hispida	Bothnian Bay	2007–2009		(Lundström *et al.*, 2014)
P. hispida	Hudson Bay	2003–2010		(Young and Ferguson, 2014)
P. hispida	Svalbard	1990; 2013		(Lowther *et al.*, 2017)
E. barbatus	Svalbard	2005–2007		(Hindell *et al.*, 2012)
H. grypus, P. vitulina	E Scotland	2003; 2005		(Matejusov *et al.*, 2008)
P. groenlandicus, C. cristata	NW Atlantic	1994–2004		(Tucker *et al.*, 2009a; Tucker *et al.*, 2009b)
H. grypus, P. hispida	Bothnian Bay	1977–1999		(Valtonen *et al.*, 2010)
E. barbatus, P. hispida, P. vitulina	Hudson Bay	1999–2006		(Young *et al.*, 2010)
H. grypus, P. vitulina	German North Sea; Baltic Sea	1994–2006	In German	(Gilles *et al.*, 2008)
H. grypus, P. vitulina	E Scotland	2005–2008		(Graham *et al.*, 2011)
H. grypus, P. vitulina	UK; Ireland		Review of diet studies published 1980-2000	(Brown *et al.*, 2012)
H. grypus, P. vitulina	Baltic Sea; Kattegat-Skagerrak	2009–2010	In Swedish	(Strömberg *et al.*, 2012)
H. grypus, P. hispida	Bothnian Bay	2008–2009		(Suuronen and Lehtonen, 2012)
C. cristata, H. grypus, P. groenlandicus	NW Atlantic	1995–2004		(Tucker *et al.*, 2013)
H. grypus, P. vitulina	E Scotland	2005–2013		(Harris *et al.*, 2014)
H. grypus, P. vitulina	SW Baltic; Kattegat	2005–2007, 2012–2013		(Pittman Botnen, 2014)
H. grypus, P. hispida, P. vitulina	Baltic Sea, Kattegat-Skagerrak	1968–2013	Review paper, including novel data	(Scharff-Olsen, 2015; Scharff-Olsen *et al.* in review.
H. grypus, P. vitulina	Scotland, England	2010–2011	Comparison of grey and harbour seal diet data	(Wilson and Hammond, 2016a)
C. cristata, P. groenlandicus	E Greenland	2008–2010		(Enoksen *et al.*, 2017; Enoksen, 2014)
H. grypus, P. hispida, P. vitulina	Baltic Sea		Review paper, including novel data	(Hansson *et al.*, 2017)

References

Aarts, G., Brasseur, S., Poos, J. J., Schop, J., Kirkwood, R., van Kooten, T., Mul, E., *et al*. 2018. Harbour seals are regaining top–down control in a coastal ecosystem. bioRxiv.

Andersen, S. M., Teilman, J., Harders, P. B., Hansen, E. H., and Hjollund, D. 2007. Diet of harbour seals and great cormorants in Limfjord, Denmark: interspecific competition and interaction with fishery. ICES J. Mar. Sci., 64: 1235–1245.

Asp, A. 2011. En pilotstudie över gråsälens (*Halichoerus grypus*) födoval runt Gotland: Kan gråsälens predation på fisk ha en strukturerande effekt på det lokala beståndet av piggvar. Examensarbete i biologi. Högskolan på Gotland: 45 pp.

Beck, C. A., Iverson, S. J., Bowen, W. D., and Blanchard, W. 2007. Sex differences in grey seal diet reflect seasonal variation in foraging behaviour and reproductive expenditure: evidence from quantitative fatty acid signature analysis. Journal of Animal Ecology, 76: 490–502.

Bowen, W. D., and Harrison, G. 2007. Seasonal and interannual variability in grey seal diets on Sable Island, eastern Scotian Shelf. NAMMCO Scientific Publications, 6: 123–134.

Brown, S. L., Bearhop, S., Harrod, C., and McDonald, R. A. 2012. A review of spatial and temporal variation in grey and common seal diet in the United Kingdom and Ireland. Journal of the Marine Biological Association of the United Kingdom, 92: 1711–1722.

Brownlow, A., Onoufriou, J., Bishop, A., Davison, N., and Thompson, D. 2016. Corkscrew Seals: Grey Seal (*Halichoerus grypus*) Infanticide and Cannibalism May Indicate the Cause of Spiral Lacerations in Seals. PLoS ONE, 11: e0156464.

de la Vega, C., Lebreton, B., Lehnert, K., Asmus, R., Siebert, U., and Asmus, H. 2018. Stable isotope composition and parasitic infections of harbor seal young-of-the-year used as prey-based diet indicators. Marine Mammal Science, 34: 7–26.

de la Vega, C., Lebreton, B., Siebert, U., Guillou, G., Das, K., Asmus, R., and Asmus, H. 2016. Seasonal Variation of Harbor Seal's Diet from the Wadden Sea in Relation to Prey Availability. PLoS ONE, 11: e0155727.

Enoksen, S. 2014. The summer diet of hooded (*Cystophora cristata*) and harp (*Pagophilus groenlandicus*) seals in the West Ice. Master thesis. The Arctic University of Norway (UiT). 51 pp.

Enoksen, S., Haug, T., Lindstrøm, U., and Nilssen, K. T. 2017. Recent summer diet of hooded *Cystophora cristata* and harp *Pagophilus groenlandicus* seals in the drift ice of the Greenland Sea. Polar Biology, 40: 931–937.

Gilles, A., Andreasen, H, Müller, S., Siebert, U. 2008. Nahrungsökologie von marinen Saeugetieren und Seevögeln für das Management von NATURA 2000 Gebieten. Teil: Marine Saeugetiere. [in German]. Final report for the German Federal Agency for Nature Conservation. 82 pp. available from https://www.bfn.de/fileadmin/MDB/documents/themen/meeresund-kuestenschutz/downloads/Forschungsberichte/NAHRUNG_Meeressaegetiere_Endbericht_2008.pdf.

Gosch, M. 2017. The diet of the grey seal [*Halichoerus grypus* (Fabricius, 1791)] in Ireland and potential interactions with commercial fisheries. PhD Thesis, University College Cork: 186 pp.

Gosch, M., Hernandez-Milian, G., Rogan, E., Jessopp, M., and Cronin, M. 2014. Grey seal diet analysis in Ireland highlights the importance of using multiple diagnostic features. Aquatic Biology, 20: 155–167.

Graham, I. M., Harris, R. N., Matejusova, I., and Middlemas, S. J. 2011. Do rogue' seals exist? Implications for seal conservation in the UK. Animal Conservation, 14: 587–598.

Granquist, S. M., and Hauksson, E. 2016. Diet of harbour seals in a salmon estuary in North-West Iceland. Icelandic Agricultural Sciences, 29: 7–19.

Hammill, M. O. 2011. Feeding of grey seals in the Southern Gulf of St Lawrence. DFO Can. Sci. Advis. Sec. Res. Doc., 2010/130: iv + 27 p.

Hammill, M. O., Stenson, G. B., Proust, F., Carter, P., and McKinnon, D. 2007. Feeding by grey seals in the Gulf of St Lawrence and around Newfoundland. NAMMCO Scientific Publications, 6: 135–152.

Hammill, M. O., Stenson, G. B., Swain, D. P., and Benoit, H. P. 2014. Feeding by grey seals on endangered stocks of Atlantic cod and white hake. Ices Journal of Marine Science, 71: 1332–1341.

Hammond, P. S., and Grellier, K. 2006. Grey seal diet composition and prey consumption in the North Sea. Final report to Department for Environment Food and Rural Affairs on project MF0319. 54 pp.

Hammond, P. S., and Harris, R. N. 2006. Grey seal diet composition and prey consumption off western Scotland and Shetland. Final report to Scottish Executive Environment and Rural Affairs Department and Scottish Natural Heritage: 41 pp.

Hammond, P. S., and Wilson, L. J. 2016. Grey Seal Diet Composition and Prey Consumption. Scottish Marine and Freshwater Science Report, 70: 47 pp.

Hanson, N., Jones Esther, L., and Harris Robert, N. 2017. Multi-decadal and ontogenetic trophic shifts inferred from stable isotope ratios of pinniped teeth. Oikos, 127: 134–146.

Hansson, S., Bergström, U., Bonsdorff, E., Härkönen, T., Jepsen, N., Kautsky, L., Lundström, K., *et al*. 2017. Competition for the fish–fish extraction from the Baltic Sea by humans, aquatic mammals, and birds. Ices Journal of Marine Science: fsx207-fsx207.

Harris, R. 2007. Assessing grey seal (*Halichoerus grypus*) diet in western Scotland. Master of Philosophy Thesis. University of St Andrews. http://hdl.handle.net/10023/432. 100 pp.

Harris, R. N., Sievers, C., and Northridge, S. 2014. Seal diet at salmon net fisheries. Seal and Salmon Research Project Report. 10 pp.

Haug, T., Falk-Petersen, S., Greenacre, M., Hop, H., Lindstrøm, U., Meier, S., Nilssen, K. T., *et al*. 2017. Trophic level and fatty acids in harp seals compared with common minke whales in the Barents Sea. Marine Biology Research, 13: 919–932.

Haug, T., Tormod Nilssen, K., Lindblom, L., and Lindstrøm, U. 2007. Diets of hooded seals (*Cystophora cristata*) in coastal waters and drift ice waters along the east coast of Greenland. Marine Biology Research, 3: 123–133.

Hindell, M. A., Lydersen, C., Hop, H., and Kovacs, K. M. 2012. Pre-Partum Diet of Adult Female Bearded Seals in Years of Contrasting Ice Conditions. PLoS ONE, 7: e38307.

ICES. 2006. Report of the Working Group on Marine Mammal Ecology (WGMME), 30 January–2 February 2006, ICES Headquarters. ACE:06. 55 pp.

ICES. 2017. Report of the Workshop on Predator–prey Interactions between Grey Seals and other marine mammals (WKPIGS), 30 April 2017, Middelfart, Denmark. ICES CM 2017/SSGEPD:18. 24 pp.

Jauniaux, T., Garigliany, M.-M., Loos, P., Bourgain, J.-L., Bouveroux, T., Coignoul, F., Haelters, J. *et al*. 2014. Bite Injuries of Grey Seals (*Halichoerus grypus*) on Harbour Porpoises (*Phocoena phocoena*). PLoS ONE, 9: e108993.

Kauhala, K., Kuunasranta, M., and Valtonen, M. 2011. Diet of grey seals in Finland in 2001–2007: a preliminary study. Hallien ravinto Suomen merialueella 2001–2007 – alustava selvitys. In Finnish with English summary. Suomen Rista, 57: 73–83.

Kavanagh, A. S., Cronin, M. A., Walton, M., and Rogan, E. 2010. Diet of the harbour seal (*Phoca vitulina vitulina*) in the west and south-west of Ireland. Journal of the Marine Biological Association of the United Kingdom, 90: 1517–1527.

Labansen, A. L., Lydersen, C., Haug, T., and Kovacs, K. M. 2007. Spring diet of ringed seals (*Phoca hispida*) from Northwestern Spitsbergen, Norway. ICES J. Mar. Sci., 64: 1246–1256.

Labansen, A. L., Lydersen, C., Levermann, N., Haug, T., and Kovacs, K. M. 2011. Diet of ringed seals (*Pusa hispida*) from Northeast Greenland. Polar Biology, 34: 227–234.

Lagström, C. 2007. Födosammansättning hos gråsäl (*Halichoerus grypus* L.) samt test av flotte för insamling av sälfekalier. Examensarbete. Naturvetenskapliga institutionen. Högskolan i Kalmar. 30 pp.

Leopold, M. F., Begeman, L., van Bleijswijk, J. D. L., Ijsseldijk, L. L., Witte, H. J., and Gröne, A. 2015. Exposing the grey seal as a major predator of harbour porpoises. Proceedings of the Royal Society B: Biological Sciences, 282: 20142429.

Lindstrøm, U., Nilssen, K., M. S. Pettersen, L., and Haug, T. 2012. Harp seal foraging behaviour during summer around Svalbard in the northern Barents Sea: Diet composition and the selection of prey.

Lowther, A. D., Fisk, A., Kovacs, K. M., and Lydersen, C. 2017. Interdecadal changes in the marine food web along the west Spitsbergen coast detected in the stable isotope composition of ringed seal (*Pusa hispida*) whiskers. Polar Biology, 40: 2027–2033.

Lundström, K., Bergenius, M., Aho, T., and Lunneryd, S. G. 2014. Födoval hos vikaresäl i Bottenviken: Rapport från den svenska forskningsjakten 2007–2009. Sveriges lantbruksuniversitet, Lysekil. Aqua reports, 2014: 23 pp.

Lundström, K., Hjerne, O., Alexandersson, K., and Karlsson, O. 2007. Estimation of grey seal (*Halichoerus grypus*) diet composition in the Baltic Sea. NAMMCO Scientific Publications, 6: 177–196.

Lundström, K., Hjerne, O., Lunneryd, S. G., and Karlsson, O. 2010. Understanding the diet composition of marine mammals: grey seals (*Halichoerus grypus*) in the Baltic Sea. Ices Journal of Marine Science, 67: 1230–1239.

Marshall, H. D., Hart, K. A., Yaskowiak, E. S., Stenson, G. B., McKinnon, D., and Perry, E. A. 2010. Molecular identification of prey in the stomach contents of Harp Seals (*Pagophilus groenlandicus*) using species-specific oligonucleotides. Molecular Ecology Resources, 10: 181–189.

Matejusov, I., Doig, F., Middlemas, S. J., Mackay, S., Douglas, A., Armstrong, J. D., Cunningham, C. O., *et al*. 2008. Using quantitative real-time PCR to detect salmonid prey in scats of grey *Halichoerus grypus* and harbour *Phoca vitulina* seals in Scotland - an experimental and field study. Journal of Applied Ecology, 45: 632–640.

Méheust, E., Alfonsi, E., Le Ménec, P., Hassani, S., and Jung, J.-L. 2015. DNA barcoding for the identification of soft remains of prey in the stomach contents of grey seals (*Halichoerus grypus*) and harbour porpoises (*Phocoena phocoena*). Marine Biology Research, 11: 385–395.

Middlemas, S. J., Barton, T. R., Armstrong, J. D., and Thompson, P. M. 2006. Functional and aggregative responses of harbour seals to changes in salmonid abundance. Proceedings of the Royal Society B-Biological Sciences, 273: 193–198.

Pittman Botnen, A. 2014. Molecular diet analysis of grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*). Master's thesis. Faculty of Science. University of Copenhagen.: 50 pp.

Ridoux, V., Spitz, J., Vincent, C., and Walton, M. J. 2007. Grey seal diet at the southern limit of its European distribution: combining dietary analyses and fatty acid profiles. J. Mar. Biol. Ass. UK, 87: 255–264.

Scharff-Olsen, C. H. 2015. Geographical variation in the diet within and between the harbour seals (*Phoca vitulina*), grey seals (*Halichoerus grypus*) and Baltic ringed seals (*Pusa hispida*) inhabiting the greater Baltic Sea. MSc thesis. University of Copenhagen. 48 pp.

Sharples, R. J., Arrizabalaga, B., and Hammond, P. S. 2009. Seals, sandeels and salmon: diet of harbour seals in St Andrews Bay and the Tay Estuary, southeast Scotland. Marine Ecology-Progress Series, 390: 265–276.

Sinisalo, T. 2007. Diet and foraging of ringed seals in relation to helminth parasite assemblages. PhD thesis. Department of Biological and Environmental Science, University of Jyväskylä, Finland. Jyväskylä studies in biological and environmental science, 174: 38 pp.

Sinisalo, T., Jones, R. I., Helle, E., and Valtonen, E. T. 2008. Changes in diets of individual Baltic ringed seals (*Phoca hispida botnica*) during their breeding season inferred from stable isotope analysis of multiple tissues. Marine Mammal Science, 24: 159–170.

Sinisalo, T., Valtonen, E. T., Helle, E., and Jones, R. I. 2006. Combining stable isotope and intestinal parasite information to evaluate dietary differences between individual ringed seals (*Phoca hispida botnica*). Canadian Journal of Zoology-Revue Canadienne De Zoologie, 84: 823–831.

Spitz, J., Dupuis, L., Becquet, V., Dubief, B., and Trites, A. W. 2015. Diet of the harbour seal *Phoca vitulina*: implication for the flatfish nursery in the Bay of Somme (English Channel, France). Aquat. Living Resour., 28: 11–19.

Spitz, J., Mariotti, L., Ridoux, V., Caillot, E., and Elder, J. F. 2010. The diet of harbour seals (*Phoca vitulina*) at the southern limit of its European distribution (Normandy, France). 2010, 8: 16.

Strömberg, A., Svärd, C., and Karlsson, O. 2012. Dietstudier av gråsäl (*Halichoerus grypus*) i Östersjön och knubbsäl (*Phoca vitulina*) i Skagerrak och Kattegatt insamlade 2010. NV-02210-11. Naturhistoriska Riksmuséet. Rapport nr 5:2012. 9 pp.

Suuronen, P., and Lehtonen, E. 2012. The role of salmonids in the diet of grey and ringed seals in the Bothnian Bay, northern Baltic Sea. Fisheries Research, 125: 283–288.

Sørlie, M. 2017. Feeding Ecology of Harbour Seals (*Phoca vitulina*) in Southern Norway. Master degree thesis. University of Agder. 37 pp.

Tucker, S., Bowen, W. D., and Iverson, S. J. 2007. Dimensions of diet segregation in grey seals *Halichoerus grypus* revealed through stable isotopes of carbon (delta C-13) and nitrogen (delta N-15). Marine Ecology-Progress Series, 339: 271–282.

Tucker, S., Bowen, W. D., and Iverson, S. J. 2008. Convergence of diet estimates derived from fatty acids and stable isotopes within individual grey seals. Marine Ecology Progress Series, 354: 267–276.

Tucker, S., Bowen, W. D., Iverson, S. J., Blanchard, W., and Stenson, G. B. 2009a. Sources of variation in diets of harp and hooded seals estimated from quantitative fatty acid signature analysis (QFASA). Marine Ecology-Progress Series, 384: 287–302.

Tucker, S., Bowen, W. D., Iverson, S. J., and Stenson, G. B. 2009b. Intrinsic and extrinsic sources of variation in the diets of harp and hooded seals revealed by fatty acid profiles. Canadian Journal of Zoology-Revue Canadienne De Zoologie, 87: 139–151.

Tucker, S., Stenson, G. B., Bowen, W. D., and Iverson, S. J. 2013. Fueling phocids: Divergent exploitation of primary energy sources and parallel ontogenetic diet switches among three species of Subarctic seals. Marine Mammal Science: 1–20.

Valtonen, E. T., Marcogliese, D. J., and Julkunen, M. 2010. Vertebrate diets derived from trophically transmitted fish parasites in the Bothnian Bay. Oecologia, 162: 139–152.

van Neer, A., Jensen, L. F., and Siebert, U. 2015. Grey seal (*Halichoerus grypus*) predation on harbour seals (*Phoca vitulina*) on the island of Helgoland, Germany. Journal of Sea Research, 97: 1–4.

Wilson, L. J. 2014. The diet and feeding ecology of harbour seals around Britain. PhD thesis. University of St Andrews.: 386 pp.

Wilson, L. J., and Hammond, P. S. 2016a. Comparing the Diet of Harbour and Grey Seals in Scotland and Eastern England. Scottish Marine and Freshwater Science Report, 7: 30 pp.

Wilson, L. J., and Hammond, P. S. 2016b. Harbour Seal Diet Composition and Diversity. Scottish Marine and Freshwater Science Report, 7: 86 pp.

Young, B. G., and Ferguson, S. H. 2013. Seasons of the ringed seal: pelagic open-water hyperphagy, benthic feeding over winter and spring fasting during molt. Wildlife Research, 40: 52–60.

Young, B. G., and Ferguson, S. H. 2014. Using stable isotopes to understand changes in ringed seal foraging ecology as a response to a warming environment. Marine Mammal Science, 30: 706–725.

Young, B. G., Loseto, L. L., and Ferguson, S. H. 2010. Diet differences among age classes of Arctic seals: evidence from stable isotope and mercury biomarkers. Polar Biology, 33: 153–162.

Zrust, M. 2017. Assessment of grey seal (*Halichoerus grypus*) diet composition in the western Baltic Sea. M.Sc. thesis. Technical University of Denmark. 98 pp.

4 ToR C. Review of additional aspects of marine mammal fishery interactions not covered by WGBY. Details of this ToR to be agreed with WGBYC

It has not yet been possible for WGMME to obtain information on what aspects of marine mammal fishery interactions WGBYC plans to cover in 2018. Thus, it has been difficult to define the frames for the treatment of this ToR. In recent years, WGMME have reviewed the development of the Bycatch Limit Algorithm Framework for determining safe bycatch limits of marine mammal species (ICES 2014) and issues related to direct impacts of seals on fisheries (ICES, 2017). In 2018, ToR B is a review of current issues in relation to indirect impacts of seals on fisheries. Here, a brief review of recent marine mammal bycatch data and mitigation measures will be presented. WGMME will be in communication with WGBYC to agree on details for ToRs to be worked on in 2019.

Belgium. In 2017, a total of 93 harbour porpoises washed ashore, a similar number as the average in the last ten years (Haelters *et al.*, 2018). Relatively important causes of death were predation by grey seals (n=11, or 32% of the animals for which a cause of death could be determined) and incidental catch (n=8, 24%). The cause of death of a white-beaked dolphin was bycatch.

For 13 out of 17 investigated seals the most probable cause of death could be assessed; three grey seals, two harbour seals and three non-identified seals had died due to bycatch. One of the harbour seals that was assessed as having been bycaught had died due to an ingested fishing hook (at least three other harbour seals were injured due to fishing hooks and might have survived).

An entangled bowhead whale, probably entangled in fishing gear, was observed close to shore. In most cases, entangled baleen whales die due to emaciation and starvation, or due to systemic infection arising from damage to tissues (Cassoff *et al.*, 2011; Barratclough *et al.*, 2014), and as such, entangled animals should be considered as lost to the population.

Denmark. In a recent paper, van Beest *et al*. (2017) used individual-based models (IBM) to predict the population-level effects of fishery time-area closure and pingers on harbour porpoise. A spatial explicit model quantified both the direct positive effects (i.e. reduced bycatch) and any indirect negative effects (i.e. reduced foraging efficiency) on the population size using the inner Danish waters as a biological system. The model incorporated empirical data on gillnet fishing effort and noise avoidance behaviour by free-ranging harbour porpoises exposed to randomized high-frequency (20 to 160 kHz) pinger signals. The IBM simulations revealed a synergistic relationship between the implementation of time-area fishing closures and pinger deployment. Time-area fishing closures reduced bycatch rates substantially but not completely. In contrast, widespread pinger deployment resulted in total mitigation of bycatch but frequent and recurrent noise avoidance behaviour in high-quality foraging habitat negatively affected individual survival and the total population size. When both bycatch mitigation measures were implemented simultaneously, the negative impact of pinger noise induced sublethal behavioural effects on the population was largely eliminated with a positive effect on the population size that was larger than when the mitigation measures were used independently.

Development of alternative fishing gear has been ongoing in Denmark in recent years by DTU Aqua (Technical University of Denmark, Institute of Aquatic Resources), aimed both at reducing bycatch of harbour porpoises and at reducing seal-inflicted damage to small-scale coastal gillnet and longline fisheries. Development of cod (*Gadus morhua*) pots in collaboration with SLU Aqua (Sweden) have led to increased target catch rates in the southern Baltic Sea, and this development work will continue in a new project to begin in mid-2018, where pots will also be loaned to interested fishermen. Trials are also ongoing with the Swedish Pontoon-trap and with small-scale Danish seine in collaboration with SLU Aqua (see below under Sweden).

DTU Aqua is developing and testing a mechanical 'pinger' aimed at reducing bycatch of harbour porpoises. Trials are also conducted with different types of AHD devices to reduce seal-inflicted damage to longline and gillnet fisheries.

Finland. In a Bayesian hierarchical analysis based on an interview survey, Vanhatalo *et al*. (2014) estimated that fisheries bycatch of grey seals in Finland, Estonia and Sweden probably exceeds 2000 animals, a substantial amount relative to the abundance of the population (ca. 30 000 animals counted, see ToR A). The reason that this bycatch rate does not have a larger impact on the population trajectory seems to be that bycatches are biased towards younger animals, which have lower survival rates and animals in poorer condition.

France. Peltier *et al*. (2016) reported that bycatches of common dolphins have been high in the Bay of Biscay and the Western Channel since the 1990s, with highest annual average of about 8000 dolphins in the period 1997–2003 and a lower annual average of about 4000 in 2008–2009. In a recent document to the IWC SC, Peltier *et al*. (2017) noted that the situation had not changed significantly since 2009. However, an unusual mass stranding of 700–800 common dolphins occurred on the French Atlantic coast (Northern Bay of Biscay) in February–March 2017. About 80% of the stranded dolphins had marks that indicated they have been bycaught in fishing gear. Assuming that only17.9% (95% CI: 9.3%–28.8%) (Peltier *et al*., 2016) of the carcasses were buoyant and subject to stranding, the total bycatch event may have included as many as 3988 common dolphins (95% CI: 2479–7677) over the period from early January to early March 2017. The time *post-mortem* was determined from Decomposition Condition Code and based on external criteria recorded for each carcass. To determine the likely geographic origin of these carcasses Peltier *et al*. (2017) used the drift model MOTHY (Daniel *et al*., 2002; 2004). In the discussion Peltier *et al*. (2017) pointed out that the most likely areas of origin corresponded to different possible fisheries among those already known to generate common dolphin bycatches (Northridge *et al*., 2006; Morizur *et al*., 2014). The sea bass (*Dicentrarchus labrax*) pair trawl fishery and a number of set-net fisheries operate widely in the coastal zone, whereas pelagic freezer trawlers and hake (*Merluccius merluccius*) bottom-set gillnet or high vertical opening trawl fisheries operate on the outer shelf and over the shelf break and slope.

Germany. In 2017, Culik *et al*. (2016) was presented to the WGMME. It described tests of the Porpoise Alerting Device (PAL), an alternative to the regular ADDs (pingers). The PAL produces aggressive click train types and emits a signal at 133 kHz with a signal strength of approx. 151 dB re 1µPa, and a repetition interval of 20s (1.2s signal length). Although most energy is emitted at 133 kHz, undertones at much lower frequencies were determined in recent recordings in the field. Potential effects of these undertones on non-target species have to be investigated. Sound propagation is less than for other pingers due to high transmission loss for high frequencies. Culik *et al*. (2016) argued that the PAL alerts the porpoises without displacing them as traditional ADDs will do. Experiments indicated that the PAL was effective in the Baltic Sea.

WGMME noted that further experiments in the North Sea showed no effect of the PAL (Culik *et al.*, 2015) and it is therefore unclear if this approach would be transferable from the Baltic Sea to other regions, or indeed if the result from the Baltic Sea is an artefact, e.g. due to small sample size. Further investigations are needed to test whether this device really leads to a better detectability of nets and an associated increase in alertness in porpoises or is only a deterrent that would add to acoustic pollution. Investigations with net-rows, PAL signals and simultaneous recordings of echolocation behaviour are needed. The effects on harbour porpoise communication signals should also be further investigated. To prevent false harbour porpoise detections at static acoustic monitoring positions due to PAL clicks, information on spatial and temporal PAL deployments have to be reported.

Despite these uncertainties, the WGMME was informed that, starting in October 2017, 1600 new PAL devices were provided by the German government for set-net fishermen as a tool to prevent incidental bycatches of harbour porpoises. The WGMME is concerned that without appropriate monitoring of the effect of the PAL (and indeed without better understanding of how the PAL affects porpoises), deployment of a large number of PALs in commercial fisheries may provide a false impression that the risks of bycatches are being reduced.

Netherlands. Brasseur (2018) reported on numbers of seals stranded dead and numbers brought into rehabilitation centres based on data from a public database on which all wildlife observations can be placed by any member of the public (www.waarneming.nl). Data are authenticated by a controller before being published.

The number of seals found dead relative to numbers of seals counted during the moult are shown in the figure below (Figure 1).

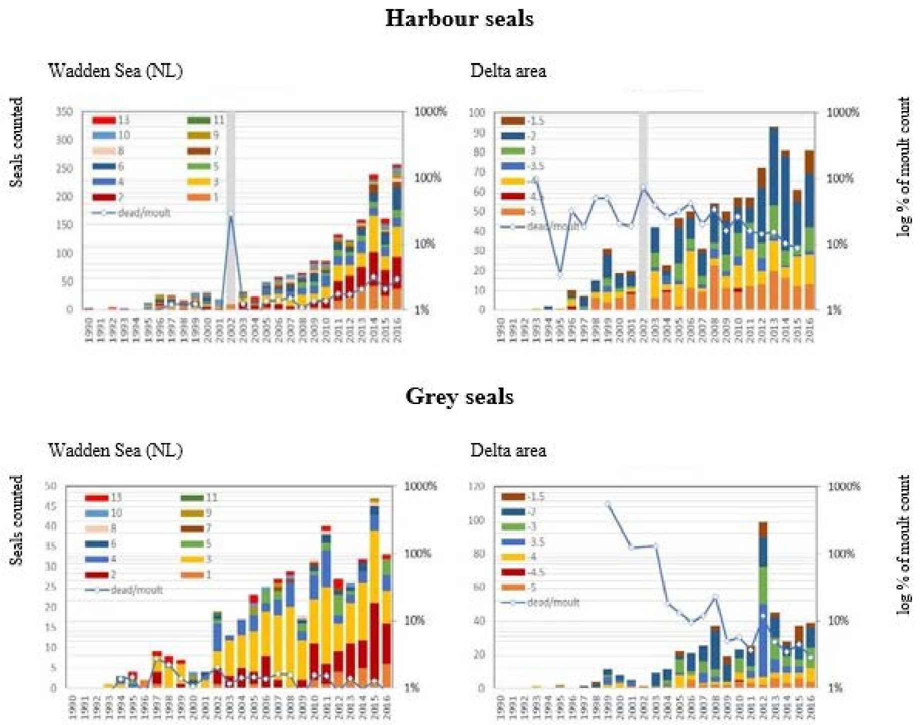

Figure 1. Annual numbers of harbour (top) and grey seals (bottom) found dead in the Dutch Wadden Sea (left) and the Delta area (right) in stacked bars for different areas (period 1990–2016). The total numbers stranded relative to the total moult counts are shown as a line (right axis; log scale). Coloured bars indicate different areas specified in Brasseur (2018).

Between 1990 and 2016 the number of animals found dead have grown only partially matching the population growth. Locally, especially in the Delta area, numbers are high, for harbour seals exceeding 10% of the total animals counted.

Given the large numbers of stranded seals reported in the Netherlands, the **WGMME support the recommendation** in Brasseur (2018) that it would be advisable to establish an official monitoring programme for seal stranding events, similar to the one that exists for cetaceans, instead of depending on data from the public. The WGMME further **recommends** that *post-mortem* examinations be made in an attempt to establish the cause of death.

Norway. Large mesh gillnets for cod and monkfish (*Lophius piscatorius*) in the Norwegian coastal zone have an annual bycatch of about 3000 harbour porpoises (Bjørge and Moan, 2016), 550 harbour and 460 grey seals (Bjørge *et al*., 2016). Bjørge reported that a small pilot study with two types of pingers was conducted in 2017 to assist in the planning of a large-scale experiment in commercial fisheries that will commence in July 2018. The Future Oceans' porpoise pinger and the Fishtek's Banana pinger were used. The Future Oceans pinger emits signals with 0.4 seconds duration at a frequency of 10 kHz and a signal strength of 132 decibels repeated every four seconds. The frequency (10 kHz) is in the audible frequency range of pinnipeds.

The Fishtek pinger emits signals with randomized intervals between 4 and 12 seconds. The signal duration is 0.4 second with a signal strength of 154 dB, and the frequency fluctuates between 50 and 120 kHz. This is outside the audible frequency range of pinnipeds.

In the cod fishery, 1723 net-weeks with pingers were compared to 2535 net-weeks without pingers. A total of eleven porpoises were caught. In nets with a pinger, one porpoise was caught every 861.5 net-weeks. In nets without pingers one porpoise was caught every 282 net-weeks. That represents a 70% reduction of the risk being bycaught in nets with pingers.

In the fishery for monkfish, 3411 net-weeks with pingers were compared with 7084 net-weeks without pingers. One porpoise was caught in nets with pingers and two in nets without pingers, resulting in no difference in catch rate. It was assumed that this was due to stochasticity with this very small sample size, a total of only three porpoises.

In nets with the Future Oceans pinger, one harbour seal was caught every 861.5 net-weeks compared to one harbour seal every 2535 net-weeks in nets without pingers. The risk of being bycaught was therefore about three times higher in nets with 10 kHz pingers. There was no difference in nets with and without the 50–120 kHz pingers.

Both the Future Oceans and the Fishtek pingers produced sounds that were outside the audible range of cod and monkfish, and should therefore have no impact on the catch rate of the target species. For monkfish no change was detected, but cod nets with pingers had 19% higher catches of cod.

Sweden. SLU Aqua (Swedish University of Agriculture Science) has several ongoing projects studying the temporal and spatial variation in harbour porpoises' presence in an area where commercial fisheries with pingers are being carried out. Preliminary results show that commercial fisheries with pingers do affect harbour porpoise presence in areas close by the fisheries. At times when fisheries are not carried out, the presence of harbour porpoises increase to the same levels as in areas where no fishing has been carried out. SLU Aqua are also trying out a newly developed pinger which emit sound with frequencies not audible to seals. Since 2015 fishermen have used pingers voluntarily in the gillnet fisheries. In 2018, up to 20 fishermen along the west and south coast will use pingers in their fisheries. SLU Aqua will supervise the project and record fishermen's logbooks when fishing with pingers.

In the Swedish small-scale coastal fisheries, alternative fishing gear have been, and is still being, developed. Pontoon traps for salmon (*Salmo salar*), whitefish (several *Coregonus* species), trout (*Salmo trutta*) and vendace (*Coregonus albula*) are used in commercial fisheries in the northern Baltic Sea. The main reason for the fishing gear development is the seal inflicted damage to fishing gear and catch, which threatens the economic viability of the gillnet fishery. Traps and pots are types of fishing gear where it is possible to protect the catch from seals. In traps and pots, the catch can be gathered in closed compartments, which in turn can be designed using a solid construction and a strong material, ensuring a public seal-safe fishing gear. At the same time, alternative gear can be developed to reduce bycatches of marine mammals.

Since 2014 there have been funding opportunities for fishermen to put forward their ideas for selective fishing gear. Projects were selected by the secretariat for selective fisheries, funded by the Swedish Agency for Water and Marine Management (SwAM) and carried out by SLU Aqua in cooperation with the involved fishermen. The purpose of the secretariat was to enable the fishing industry to develop selective fishing gear to enhance the transit to the new landing obligation. Since 2015 there has been several projects developing selective fishing gear such as trapnet fisheries for cod, multifunctional pots fishing for cod and lobster (*Homarus gammarus* and *Nephrops norvegicus*), and pots for cod.

Sweden and Denmark have been cooperating in developing cod pots as an alternative to the gillnet cod fisheries in the southern Baltic Sea. During recent years the development of codpots has led to an implementation project where several fishermen can loan and try out codpots voluntarily. Today there are a few fishermen using pots in the commercial fishery for cod in the Baltic Sea.

A more sustainable method relative to trawling is bottom seine netting, such as the Danish bottom seine. Bottom seines are generally considered less damaging than bottom trawls (ICES, 2006) and well-managed seine fisheries generally have minor ecosystem impacts (Morgan and Chuenpabgdee, 2003). In 2015 SLU Aqua started to develop a seine net modified for small open boats and tried it out for pelagic and demersal species as substitute for gillnet. The small-scale seine net has shown to yield commercial catches for benthic species such as vendace, white-fish and flounder (several species). However, the fishing gear is still under development and modifications are needed to get the fishing gear to work properly. The study also showed that to generate high catches, dedicated and trained fishermen with plenty of local knowledge of the fish and the bottom structure in the area, is needed. A new project in cooperation with the Swedish Fishermen's Producing Organisation will be carried out in 2018 involving the development of a seine net for cod fisheries.

4.1 References

Barratclough, A., Jepson, P.D., Hamilton, P.K., Miller, C.A., Wilson, K. and Moore, M. J. 2014. How much does a swimming, underweight, entangled right whale (*Eubalaena glacialis*) weigh? Calculating the weight at sea, to facilitate accurate dosing of sedatives to enable disentanglement. Marine Mammal Science 30(4): 1589–1599.

Bjørge, A. and Moan, A. 2016. Revised estimates of harbour porpoise (*Phocoena phocoena*) bycatches in two Norwegian coastal gillnet fisheries. Report to the Scientific Committee of the International Whaling Commission, Bled, Slovenia, June 2016. SC/66b/SM/03. 16pp.

Bjørge, A., Moan, A., Nilssen, K.T. and Øigård, T.A. 2016. Bycatch of harbour and grey seals in Norway. Report to the NAMMCO working group on coastal seals. Reykjavik, Iceland. March 2016. SC/23/CSWG/08. 15pp.

Brasseur, S. 2018. Stranding and Rehabilitation in Numbers: Population development and stranding data on the Dutch coasts 1990–2016; Analysis of new data from a public database. Den Helder, Wageningen Marine Research, Wageningen Marine Research report C108/17. 36 pp. available from http://edepot.wur.nl/440805.

Cassoff, R.M., Moore, K.M., McLellan, W.A., Barco, S.G., Rotstein, R.D.S. and Moore M.J. 2011. Lethal entanglement in baleen whales. Diseases of Aquatic Organisms 96: 175–185.

Culik, B., Conrad, M., Tobias Schaffeld, T., Christian von Dorrien, C. and Lotte Kindt-Larsen, L. 2015. Deploying porpoise alerting device (PAL) in Baltic and North Sea gillnet fisheries. In: Wright, A.J., and Robertson, F.C. (eds). New mitigation methods and evolving acoustics exposure guidelines. Report from the European Cetacean Society Conference Workshop, St Julian, Malta, European Cetacean Society Special Publication Series No. 59, 74 pp.

Culik, B., von Dorrien, C., and Conrad, M. 2016. Porpoise Alerting Device (PAL): synthetic harbour porpoise (*Phocoena phocoena*) communication signals influence behaviour and reduce by-catch. BfN Skripten, 451, 150–155. Retrieved from https://www.bfn.de/fileadmin/BfN/service/Dokumente/skripten/Skript451.pdf.

Daniel, P., Josse, P., Dandin, P., Lefevre, J., Lery, G., Cabioc'h, F., Gouriou, V. 2004. Forecasting the Prestige oil spills. Interspill Conference, 14–17 June 2004, Trondheim, Norway.

Daniel, P., Jan, G., Cabioc'h, F., Landau, Y., Loiseau, E. 2002. Drift modelling of cargo containers. *Spill Science Technology Bulletin* **7**(5-6): 279–288.

Goldstein, T., Mazet, J.a.K., Gulland, F.M.D., Rowles, T., Harvey, J.T., Allen, S.G., King, D.P., Aldridge, B.M. and Stott, J.L. 2004. The transmission of phocine herpesvirus-1 in rehabilitating and free-ranging Pacific harbor seals (*Phoca vitulina*) in California. *Veterinary Microbiology* **103**: 131–141.

Haelters, J., Kerckhof, F., Moreau, K., Potin, M., Doom, M. and Jauniaux, T. 2018. In press. Strandingen en waarnemingen van zeezoogdieren en opmerkelijke vissen in België in 2017 [Strandings and sightings of marine mammals and some remarkable fish species in Belgium in 2017]. Koninklijk Belgisch Instituut voor Natuurwetenschappen (KBIN), Brussels, Belgium.

ICES. 2014. Report of the Working Group on Marine Mammal Ecology (WGMME), 10–13 March 2014, Woods Hole, Massachusetts, USA. ICES CM 2014/ACOM:27. 234 pp.

ICES. 2017. Report of the Working Group on Marine Mammal Ecology (WGMME), 6– 9 February 2017, St Andrews, Scotland, UK. ICES CM 2017/ACOM:27. 103 pp.

Jensen, L.F., Ejbye-Ernst, R., Michaelsen, T.Y., Jensen, A., Hansen, D.M., Nielsen, M.E. and Pertoldi, C. 2017. Assessing the genetic effects of rehabilitating harbor seals (*Phoca vitulina*) in the Wadden Sea using stochastic simulations. Mammal Research 62: 363–372.

Morgan, L.E. and Chuenpabgdee, R. 2003. Shifting gears. Addressing the Collateral Impacts of Fishing Methods in US Waters. *PEW Science Series*. Island Press, Washington, D.C. 48 pp.

Morizur, Y., Gaudou, O. and Demaneche, S. 2014. Analyse des captures accidentelles de mammifères marins dans les pêcheries françaises aux filets fixes. Ifremer RBE/STH/LBH/2014. http://archimer.ifremer.fr/doc/00209/32016/.

Northridge, S., Morizur, Y., Souami, Y. and Van Canneyt, O. 2006. PETRACET: Project EC/FISH/2003/09. Final report to the European Commission 1735R07D, June 2006, 29pp.

Peltier, H., Authier, M., Deaville, R., Dabin, W., Jepson, P.D., Van Canneyt, O., Daniel, P., and Ridoux, V. 2016. Small cetacean bycatch as estimated from stranding schemes: The common dolphin case in the Northeast Atlantic. *Environmental Science & Policy* **63**: 7–18.

Peltier, H., van Canney, O., Dabin, W., Dars, C., Demaret, F. and Ridoux V. 2017. New fishery-related unusual mortality and stranding events of common dolphins in the Bay of Biscay, February–March 2017, France. Document to the IWC Scientific Committee 2017 (HIMWP8). 6 pp.

Stamper, M.A., Gulland, F.M.D. and Spraker, T. 1998. Leptospirosis in rehabilitated Pacific harbor seals from California. *Journal of wildlife diseases* **34**: 407–410.

Vanhatalo, J., Vetemaa, M., Herrero, A., Aho, T., Tiilikainen, R. 2014. By-Catch of Grey Seals (*Halichoerus grypus*) in Baltic Fisheries-A Bayesian Analysis of Interview Survey. PLoS One 9(11): e113836. https://doi.org/10.1371/journal.pone.0113836.

van Beest, F.L., Kindt-Larsen,L., Bastardie, F., Bartolino, V. and Nabe-Nielsen, J. 2017. Predicting the population-level impact of mitigating harbor porpoise bycatch with pingers and time-area fishing closures. Ecosphere 8(4): e01785.10.1002/ecs2.1785.

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). At that time, 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) (ICES, 2008). Despite attempts, a unified database could not be finalised and requirements with regard to seal numbers are met by presenting a table with only the latest counts for each area.

In 2015, in a separate effort, OSPAR issued a formal data call to its Contracting Parties to submit data to support the assessment of the EU MSFD common indicators for seals: M-5 (grey seal pup production) and M-3 ((harbour and grey) seal population abundance and distribution). These data formed the basis of draft assessments of indicators M-5 and M-3 for OSPAR's Intermediate Assessment (OSPAR 2017). However, data were rarely submitted in the requested format and occasionally had to be gleaned from literature and Internet sources. These data constitute the current OSPAR database, which is to be replaced and/or supplemented with data obtained from a 2018 data call.

Also, 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).

5.2 The 'ICES/WGMME seal database'

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 Norway, Sweden, Denmark, Germany, the Netherlands, Belgium, France, UK 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, 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.

For a few areas, numbers are available between 1986 and 2014 in a WGMME seal database. However, the information is far from complete and up to 2017 the ToR (a) requirement has been fulfilled by presenting a table with only the latest counts for each area. For most countries and years there are no data, either because the database has not been updated or because annual surveys were not performed. Considerable effort would be necessary to update this database with information from each country listed, but also to provide data in a similar format allowing for comparison. In some cases, population estimates are made based on (partial) counts, while in others rough counts are presented. Also, surveys may be timed in different seasons. Thus, although a significant amount of processed data is publicly available from many areas (e.g. UK, Wadden Sea), these do not necessarily feed directly into one unified format.

5.3 OSPAR seal database'

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 OSPAR intermediate MSFD assessments were performed at the scale of Assessment Units defined separately for harbour and grey seals and are summarised 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 haul-out 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. 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, but restricted, at: http://ices.dk/marine-data/data-portals/Pages/Biodiversity.aspx]

5.4 Future database concerns

As in 2017, the WG discussed whether it is necessary to maintain two seal databases and if the more recently collated OSPAR database would suffice. It should be noted that, first to ensure correct interpretation, data collected at the resolution used for the MSFD can only be produced by the bodies responsible for the collection of the data themselves, while the resolution obtained is not necessary for a more general inspection as is done in ICES/WGMME. Second, the area covered by the OSPAR database overlaps only partially with the area covered by the ICES/WGMME database.

Discussions both within the WGMME and between this working group and ICES HQ resulted in the following solutions:

- The more detailed OSPAR database covering European waters will continue to be updated only via the formal OSPAR data call procedure, by national coordinators in each Contracting Party. WGMME will support ICES and OSPAR with guidelines for contracting parties regarding data submission for the 2018 data call.
- to provide for an overview of the status of the seal species concerned, the ICES/WGMME database will be maintained at a less detailed, but geographically broader level (e.g. including Iceland, Canada, USA). 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 (see ToR A, this report, for trajectories). 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, so the data are only edited by the WGMME. Copies of the database can be freely shared with interested parties. WG decided to use a less ambitious format for the database with an emphasis on time-series of abundance or count data on a management unit basis. Such data would largely fulfil the requirements for the standing ToR A (Review and report on any new information on seal and cetacean population abundance, population/stock structure) of WGMME. In this format other seal species, such as harp, hooded and ringed could potentially be added.

5.5 Recommendation

The **WGMME supports the format** suggested by ICES and OSPAR for collecting seal abundance and distribution data and will assist in developing guidelines for contracting parties for the submission of data under this format.

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. 2016. Report of the Working Group on Marine Mammal Ecology (WGMME). ICES, Madrid, Spain. ICES CM 2016/ACOM:26. 118 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.

OSPAR 2017. Seal abundance and distribution, Intermediate Assessment 2017. https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/biodiversity-status/marine-mammals/seal-abundance-and-distribution/

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Annex 2: Draft Terms of Reference for 2019

2018/X/ACOMXX The **Working Group on Marine Mammal Ecology** (WGMME), chaired by Anders Galatius (Denmark) and Anita Gilles (Germany), will meet in Büsum, Germany, X–X February 2019 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 and update information on the ecological role of marine mammals, e.g. influence on structure, function and transfer of energy (and of parasites) in marine foodwebs;

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

d) Review the population-level effect of cumulative human impacts on marine mammals and further develop and/or update the threats matrix;

e) 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 the ecological role of marine mammals, in response to emerging issues e.g. in seal-fisheries interactions. The group also proposes to include a review on the occurrence of trematode gastric parasites and the role of fish in the life cycles of respiratory parasites of marine mammals to follow up other areas not covered by this year´s review of digestive tract parasites.

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 aims to address the interactive effects of multiple stressors (e.g. noise, fisheries, marine constructions, pollution, habitat degradation).

ToR e is a standing term of reference to keep the reworked seal database up to date.

WGMME will report by x.x.2019 for the attention of the Advisory Committee.