WORKING GROUP ON MARINE MAMMAL ECOLOGY (WGMME)

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i Executive summary

The Working Group on Marine Mammal Ecology (WGMME), chaired by Anita Gilles (Germany) and Anders Galatius (Denmark), met at the Institute for Terrestrial and Aquatic Wildlife Research, University of Veterinary Medicine Hannover, Foundation, in Büsum, Germany, during 11–14 February 2019. On behalf of the working group, the chairs would like to thank the Institute for Terrestrial and Aquatic Wildlife Research (ITAW) for hosting the meeting.

A total of 17 participants from eight countries attended the meeting on site. The list of participants, including those contributing remotely, and contact details are given in Annex 1. 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 could all be addressed. The Working Group gratefully acknowledges the support given by several additional experts that kindly provided information and/or reports for use by WGMME and evaluated the threats matrices and related text (see Annex 1).

Two terms of references were standing ToRs; 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. Usually data on marine mammal surveys are only conducted during one season, but new data from Ireland revealed profound differences between summer and winter abundance and distribution of cetaceans, with implications for the spatial management of these species. New habitat modelling for cetacean species in the Northwest European Seas and North Atlantic Ocean and Mediterranean based on collated survey data as well as findings on threats to marine mammals such as bycatch, pollution, marine debris and noise were summarised under this ToR.

Under ToR B, WGMME reviewed information on the ecological roles of marine mammals, expanding on efforts to review impacts on fisheries, interspecific competition, diet and impact on fish stocks from previous reports. The complexity and multitude of ecological interactions is underlined. A special emphasis under this ToR was put on the role of marine mammals as hosts and vectors of parasites.

ToR C was implemented to review aspects of marine mammal fishery interactions not covered by ICES WGBYC. In 2019, WGMME focused its efforts on i) reviewing seal interactions with fisheries, including the numbers of bycaught seals, ii) investigating additional data sources for cetacean bycatch such as strandings, voluntary recording schemes and interview surveys, iii) identifying of potential bycatch risk areas, and iv) a review of recent work on setting safe limit thresholds for bycatch of marine mammals. To implement bycatch data in the management of marine mammals it is stressed that they should be aggregated on a management unit basis and corresponding data on abundance should be available.

Under ToR D, WGMME followed up and updated on the 2015 threat matrix for the main marine mammal species in each of the regional sea areas. In addition to this update, a review of cumulative impacts from multiple environmental pressures and recent efforts to meaningfully analyse such cumulative impacts was produced. The approaches used to assess cumulative impacts are usually limited to a particular pressure. Efforts to expand the scopes of these tools are impeded by the complexity of the relevant interactions and knowledge gaps.

ToR E, updating the database for seals, is the second standing term of reference. This year, WGMME adapted the ICES WGMME seal database to a proper database format and updated the data. WGMME also repeated its support for a data call from OSPAR to provide data for assessments under OSPAR indicators M3 and M5 on seal abundance and distribution.

The WGMME updated ToRs for 2020 (see Annex 2) and discussed meeting venues. The Institute of Marine Sciences (ICM-CSIC) offered to host the 2020 meeting in Barcelona (Spain) alongside the meeting of ICES WGBIODIV in order to arrange common sessions and establish cooperation between the two groups (see proposed ToR B for 2020).

ii Expert group information

Expert group name	Working Group on Marine Mammal Ecology (WGMME)
Expert group cycle	Annual
Year cycle started	2019
Reporting year in cycle	1/1
Chair(s)	Anita Gilles, Germany
	Anders Galatius, Denmark
Meeting venue(s) and dates	11–14 February 2019, Büsum, Germany, (17 participants)

iii Term of reference

Term of reference	Addressed in this report
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;	Yes
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;	Yes
c) Review additional aspects of marine mammal fishery interactions not covered by WGBYC. Details of this ToR to be agreed with WGBYC;	Yes
d) Review the population-level effect of cumulative human impacts on marine mammals and further develop and/or update the threats matrix;	Yes
e) Update the database for seals.	Yes

1 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.

1.1 New abundance information

1.1.1 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, 2018
	North of 62N	2015	3872		
	South of 62N	2011–2016	1128		
	Finmark	2012–2013	981		
	Skagerrak	2016–2018	880		
Iceland		2016	7652		Thorbjörnsson *et al.*, 2017
Wadden Sea		2018	27 500 estimated	9285 (2018)	Galatius *et al.*, 2018 areas were missed during the moult count
Dutch Delta Area		2016/2017	868 (2016)	84 (2017)	Arts *et al.*, 2018
France		2018	1088	198	S. Poncet, 2018
UK					
	Scotland	2013–2017	26 553		SCOS, 2018
	England and Wales	2015–2016	5092		SCOS, 2018
	Northern Ireland	2011	948		SCOS, 2018
Ireland		2011–2012	3489		Duck and Morris, 2013
USA		2012	75 834		Waring *et al.*, 2015
Canada					NAMMCO
	south of Labrador	1970s	12 700		
	Estuary and Gulf of St Lawrence	1994–2000	4000–5000		
Sweden and Denmark					
	Skagerrak	2016	6240		Swedish Museum of Nat. Hist., Markus Ahola
	Kattegat/ Danish Straits	2016 (pups: 2017)	9394	2331 (only counted in Danish area)	Swedish Museum of Nat. Hist., Galatius *et al.*, 2019
	southern Baltic	2017	950		Galatius *et al.*, 2019
	Limfjord	2017	1100	268	Galatius *et al.*, 2019
	Kalmarsund	2016	1100		HELCOM

Table 2. Recent grey seal survey data.

Country		Recent Survey Year(s)	Adults (moult)	Pups	References
Norway	Tomso & Finmark	2015–2016		271	Nilssen and Bjørge, 2017a & b
	Norway north of 62N	2014–2015		398	Nilssen and Bjørge, 2017a & b
	Norway south of 62N	2017		40	Nilssen and Bjørge, 2017a & b
Iceland		2017	6269	1452	Granquist, S.M. and Hauksson, E. 2019
Wadden Sea		2017	5445	1279	Brasseur *et al.*, 2017
Dutch Delta Area		2017	1489		Arts *et al.*, 2018
France		2016	895	43	Vincent *et al.* (in revision)
UK	Inner Hebrides	2016		4541	SCOS, 2018
	Outer Hebrides	2016		15 732	SCOS, 2018
	NW Scotland	2016		706	SCOS, 2018
	Scottish North Sea	2016, 2004*		33 177	SCOS, 2018; * Shetland
	English North Sea	2017		8689	National Trust, Lincolnshire Wildlife Trust, Natural England, Friends of Horsey Seals
	SW England & Wales	2016		2000	SCOS, 2018
Republic		2012		2100	Ó Cadhla *et al.*, 2013

of Ireland					
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	2019	30 000		HELCOM

Table 3. Recent ringed seal survey data.

Country		Survey Year(s)	Adults (moult)	Pups	References
Sweden, Finland	Bothnian Bay	2018	9919		HELCOM (normal ice conditions)
	Bothnian Bay	2015	19 936		HELCOM (unusual ice conditions)
Estonia, Finland, Russia	Gulf of Finland	2018	95 + 13		M. Verevkin, 2018 (Russian side: 95, average taken of range, Finnish side: 13, all animals observed in complete survey, Estonia: remaining ice was observed, but no seals were observed)
Estonia, Latvia	Gulf of Riga	2018	1152		I. Jüssi, 2018
Finland	Finnish Archipelago Sea	2018	122 observed, population estimate 200–300		M. Kunnasranta, 2018

1.1.1.1 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 (Figure 1). 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 2017 indicated a population abundance of 6269 animals, based on a pup survey yielding 1452 pups (Figure 2; Granquist and Hauksson 2019).

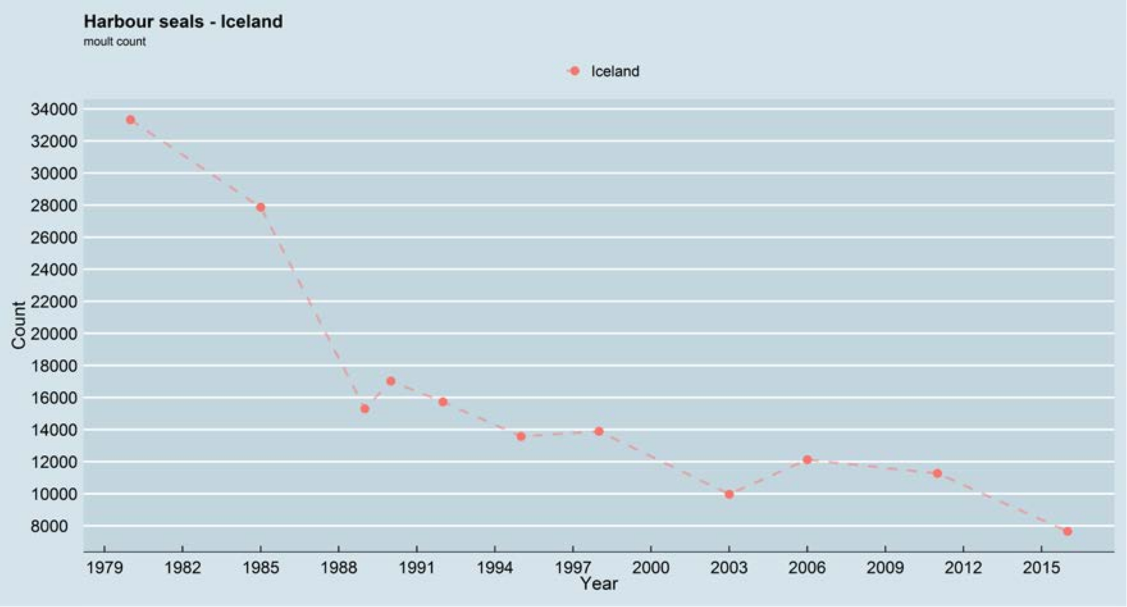


Figure 1. The trend of survey results of harbour seals in Iceland.

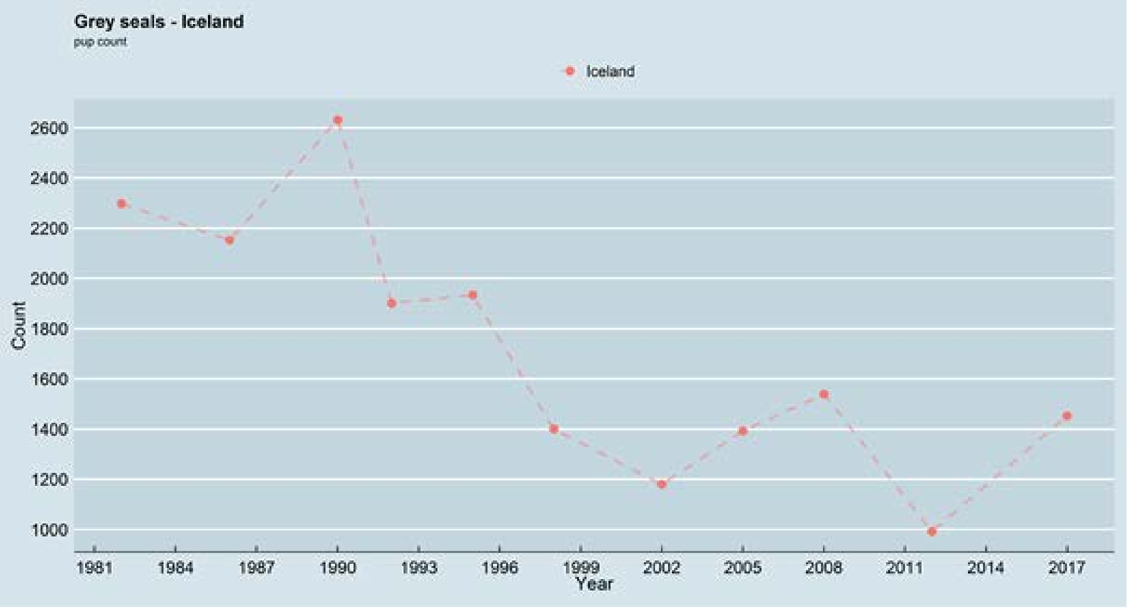


Figure 2. The trend of counted grey seal pups in Iceland.

1.1.1.2 Baltic Sea

Ringed seals

Ringed seal (*Pusa 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 to some extent every year in the Bothnian Bay, where the surveys have been carried out since 1988. The number of hauled out individuals during the surveys in normal ice-conditions has increased from the level of around 2000 in the first survey years to 9919 in 2018 (Figure 3), corresponding to an annual average population increase being 4.7% per year. The increase rate has been slightly higher in the latter half of the period (2004–2018: 5.6% per year). Both increase rates are clearly below the intrinsic natural rate in a situation without apparent natural limiting factors. Anomalous survey results in 2013, 2014, 2015 and 2017 with early ice-breakup have been excluded from the trend analysis, as they are not comparable to previous data. The phenomenon behind the anomalous results and the role of early ice-breakup are not fully understood. This was discussed in the WGMME 2018 report (ICES, 2018).

The ringed seal subpopulation in the Bothnian Bay is the largest in the Baltic. It has recovered from the hunting-derived population decline during the 20th century and subsequent reproductive problems caused by environmental contaminants. However, recently raised hunting quotas together with decreasing ice-conditions increase the pressures on this subpopulation.

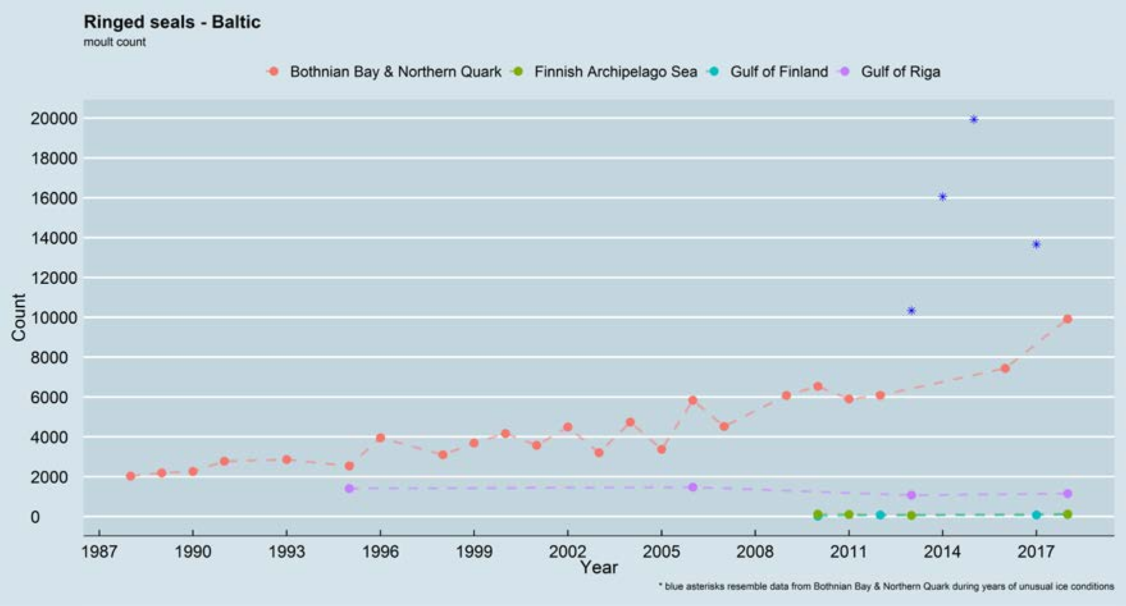


Figure 3. Trends of estimated numbers of ringed seals hauled out on sea ice during moult surveys in the Baltic.

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.

In addition to the largest subpopulation in the Bothnian Bay, Baltic ringed seal subpopulations can be found in the Gulf of Riga, the Finnish Archipelago Sea and the Gulf of Finland. These subpopulations are threatened with extinction, probably as a result of reduced breeding success caused by reduced extent and duration of sea ice with less snow compared to historically average winters. This was covered in more detail in the WGMME 2018 report (ICES, 2018).

While the warmer winters have recently challenged 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 status of the southern ringed seals as well as the roles of climate warming and other factors on them was discussed in more detail in the WGMME 2018 report (ICES, 2018).

In 2018, ice conditions enabled aerial surveys for southern subpopulations during moulting time. In the area occupied by the Gulf of Riga subpopulation, ice only occurred in relatively small scattered areas by the Saaremaa and Hiiumaa islands. During an aerial survey on 13th of April, the entire ice covered area was observed. The result of 1152 ringed seals represent the hauled out population (excluding the individuals in the water during the survey). The result is in line with the recent successful surveys (Figure 3) under both over sea ice (2013: 1077 ± 449) and in ice-free conditions (2008: 1055, 2014: 1010 and 2016: 834).

In the Finnish Southwestern Archipelago, two surveys were carried out on 10th and 11th of April 2018. During the later day, 122 animals were seen. The result is slightly lower than the maximum results of ca. 140 in 2010 and 2011 (Figure 3) whereas the count of 67 animals on 10th of April 2018 was close to the lower results of ca. 60 animals from both surveys of 2013. In the Finnish Southwestern Archipelago, the entire ice covered area was observed, though some proportion of the animals will be in the water, and some may be hauling out on land in ice-free parts of the area and a few may have been missed behind islands.

On the Finnish side of the Gulf of Finland, two surveys covering the entire ice covered area were carried out in 12th and 13th of April, and 9 and 13 ringed seals were observed, respectively. On the Russian side, two surveys were conducted in 14th and 18th of April, the survey strips covering 14 and 13% of the sea ice, respectively. From the ten and 13 seals, the estimates of the hauled-out population are 64–76 and 89–101, respectively. One aerial survey was conducted over limited floating ice sheets in Estonian part of Gulf of Finland, too, where no seals were observed. The haul out distribution is concentrated to the northeastern parts of the gulf, where the most stable fast-ice is located. The combined estimated Russian and observed Finnish hauled out numbers are shown in Figure 3.

Grey seals

Monitoring of the grey seal population in the Baltic Sea (*Halichoerus grypus* ssp. grypus) 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 are used for assessing abundance and trends. 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 in the whole Baltic Sea has been at the level of 30 000 animals during recent years (HELCOM, unpublished). Of the hauled-out population, over 80% is found in the core moulting area in 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. As the abundance of the population has increased, its range has expanded to also include the southern Baltic, where grey seals have been breeding regularly, although in small numbers, since 2003 (Galatius *et al.*, 2019). In recent years, pups are now also annually observed in Kattegat. (Galatius *et al.*, 2019). This expansion has brought Baltic grey seals in contact with the Atlantic subspecies, and there are strong indications for hybridisation between the two groups based on microsatellite data from the southern Baltic (Fietz *et al.*, 2016). The annual numbers of grey seals observed during moult surveys in different subareas of the Baltic are shown in Figure 4.

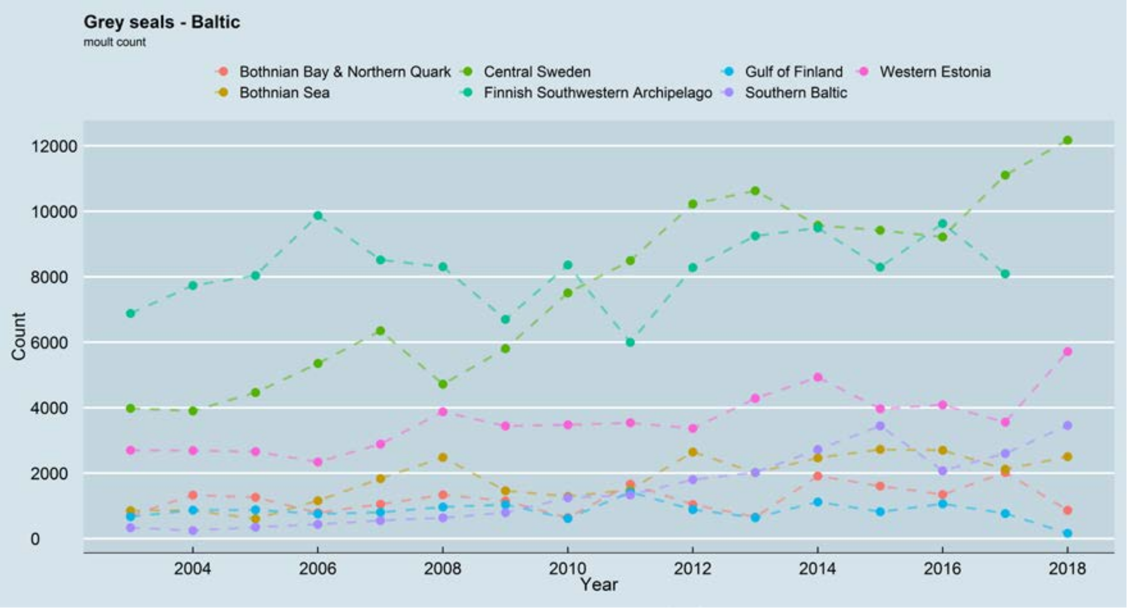


Figure 4. Trends for results of moult counts of grey seals in subareas of the Baltic Sea.

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 (Figure 5). They are split into the four management units: Limfjord, Kattegat and the Danish Belt Sea, 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 2018, 950 seals were counted in the fjord (Aarhus University).

KATTEGAT AND THE DANISH BELT SEA: 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 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. Hauled-out population estimate was 9400 in 2016 (HELCOM, 2018). Average of the two highest survey results out of three has been used for the estimate. Since 2017 only two surveys have been carried out in Danish side, a new measure for the hauled-out population will be developed.

SOUTHWESTERN BALTIC: Southwestern Baltic harbour seals were also hit hard by the PDV epidemics of 1988 and 2002. 950 seals were counted in the area in 2017 (Galatius *et al.*, 2019). Since the 2002 and until 2011, the population grew with an average annual rate of 13%. From 2011 to 2017, this rate has been reduced to 8%. It is unclear whether this reduced growth stems from harbour seals approaching carrying capacity, increased presence of grey seals in the area or a combination.

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, 2018).

Figure 5. Trends of moult counts of harbour seals in the Kattegat and the Danish Belt Sea, Southwestern Baltic, Limfjord and Kalmarsund.

1.1.1.3 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 latter 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 6).

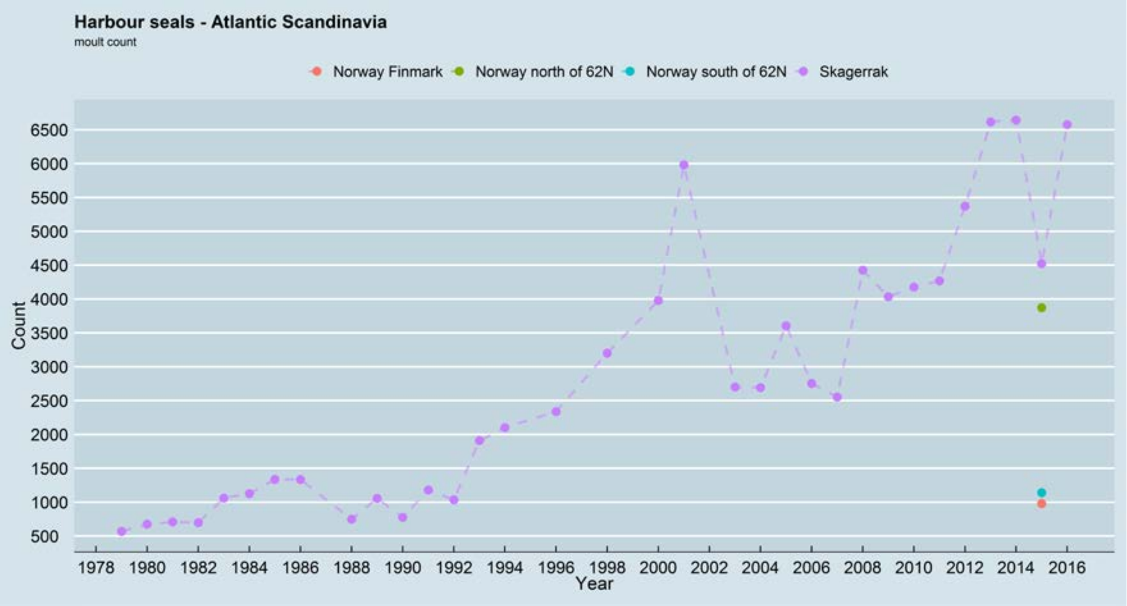


Figure 6. Trends of moult counts of harbour seals in the Skagerrak.

1.1.1.4 Continental coast, Wadden Sea to France

Harbour seals

WADDEN SEA (Denmark, Germany, the Netherlands): 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 2018, a hauled-out population count of almost 27 500 harbour seals were estimated as part of the Wadden sea area could not be surveyed during the moult (Galatius *et al.*, 2018).

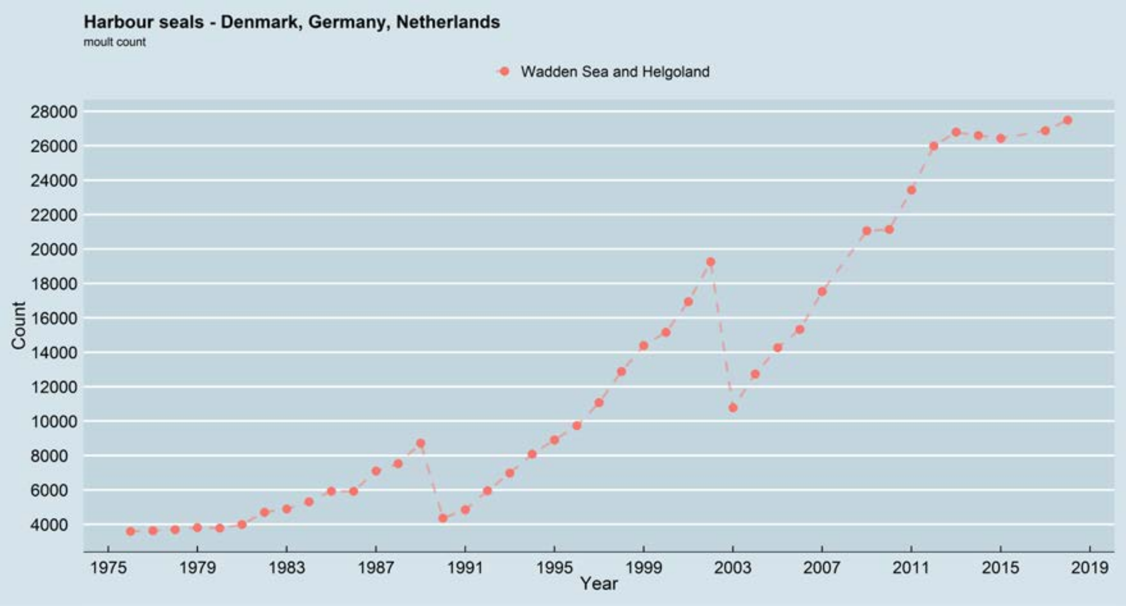


Figure 7. Trends of counts of moulting harbour seals in the Wadden Sea.

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 (84 pups in 2017) to explain its growth. Telemetry data show regular exchange between the areas. Over 850 animals were counted in the Dutch Delta area in 2016 (Arts *et al.*, 2018), 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 2018, seal counts amounted to almost 1100 harbour seals in the colonies on the coasts of Brittany and Normandy (Data compiled by Poncet S.; data owners: PNMI (AFB), SYMEL CDL, ADN, GDEAM-62, GMN, Bretagne Vivante, Picardie Nature, Réserve naturelle nationale du Domaine de Beauguillot (PNRMCB), Syndicat Mixte Baie du Mont-Saint-Michel, PNMI (AFB), Maison de l'estuaire, AFB, CMNF, Bretagne Vivante, ONCFS, RNN des Sept-Iles / LPO, ONCFS).

In Belgium there are no true seal colonies, however tens of animals strand annually along the coasts. The number of harbour seals observed hauling out in Belgium, especially in the port of Nieuwpoort, is rising and seals are seen daily. In 2018, up to 15 animals were observed hauled out (for Belgian standards, unprecedented numbers). In 2017, at least four animals were injured by fishing hooks, with at least one animal dying due to ingestion of fishing hooks, while in 2018 at least three animals were injured (Haelters *et al.*, 2018). (Also see ToRs B & C).

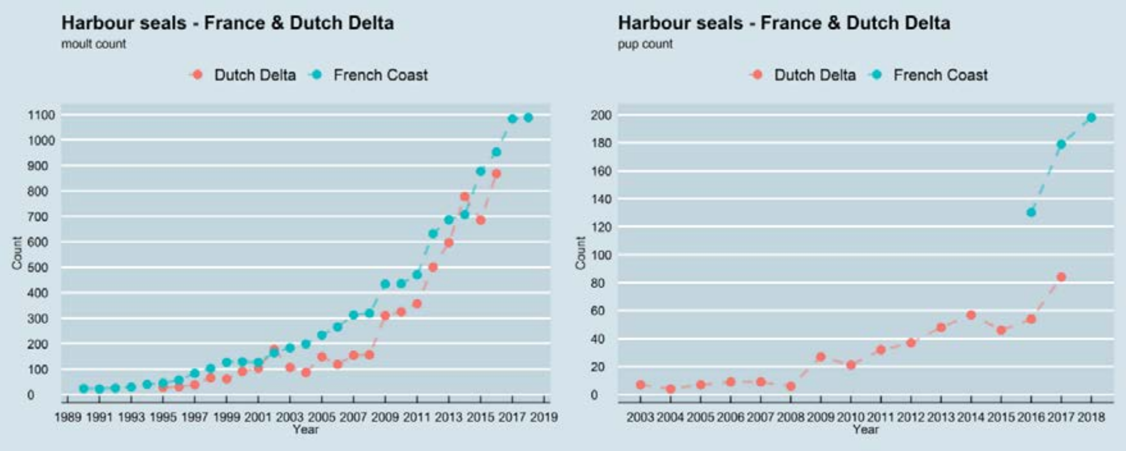


Figure 8. Trends of counts of moulting harbour seals and harbour seal pups in the Dutch Delta and French Coast.

Grey seals

After centuries of practical absence, grey seals have shown a remarkable recovery in the Wadden Sea area (Figure 9). Partially fuelled by immigration from the UK (Brasseur *et al.*, 2015), colonies started in Germany and the Netherlands and have since expanded 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, the Wadden Sea and the UK, where numbers are growing. In 2017, a maximum count of 1489 grey seals was reported (Arts *et al.*, 2018). In France, there are also breeding colonies, and numerous exchanges with the UK and the Wadden Sea have also been recorded with telemetry. The maximum count along the French coasts amounted to 1070 in 2018, and on the breeding sites, 50 pups were observed (Figure 10). (Data compiled by Poncet S.; data owners: PNMI (AFB), SYMEL CDL, ADN, GDEAM-62, GMN, Bretagne Vivante, Picardie Nature, Réserve naturelle nationale du Domaine de Beauguillot (PNRMCB), Syndicat Mixte Baie du Mont-Saint-Michel, PNMI (AFB), Maison de l'estuaire, AFB, CMNF, Bretagne Vivante, ONCFS, RNN des Sept-Iles / LPO, ONCFS).

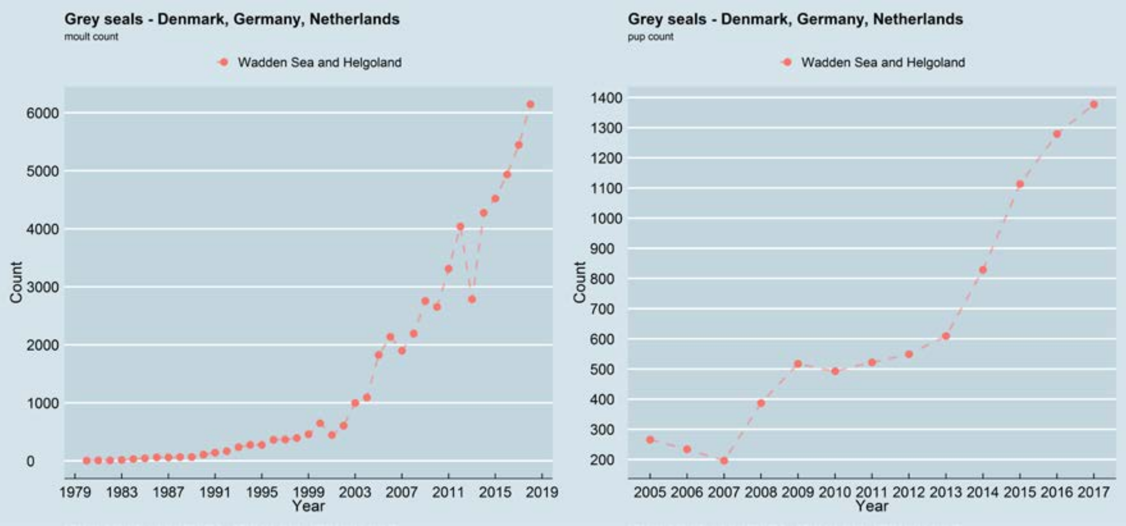


Figure 9. Trends of moulting grey seals and grey seal pups in the Wadden Sea.

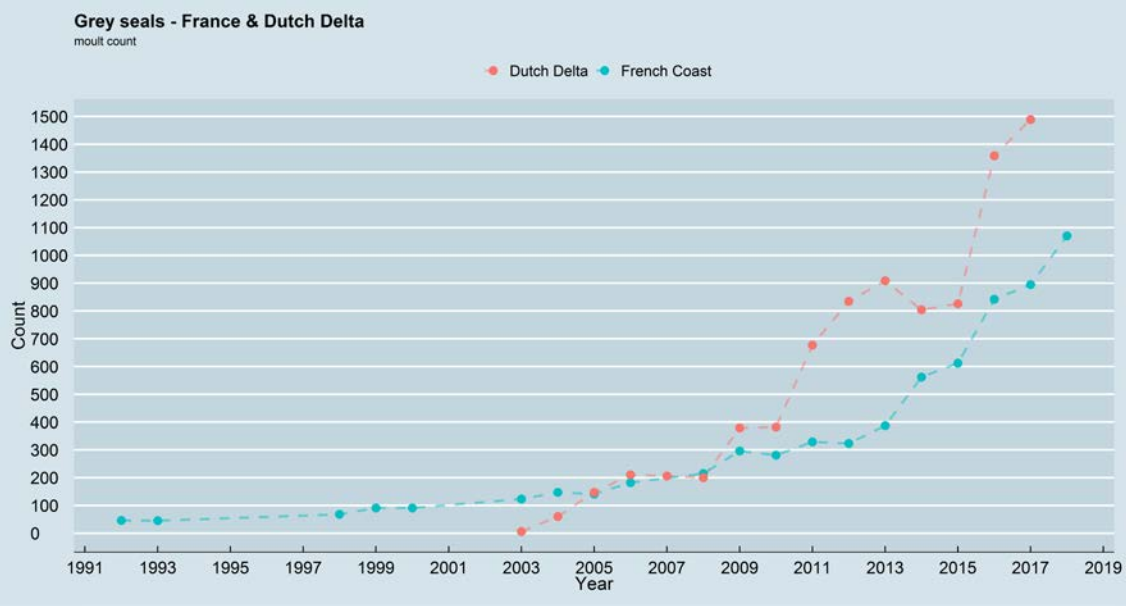


Figure 10. Trends of moult counts of grey seals in the Dutch Delta Area and France.

1.1.1.5 UK

Harbour seal

Harbour seals in the UK are counted annually during the moult period, although not all haul-outs are surveyed annually. Therefore, data are collated for multiyear survey periods during which all large haul-outs are surveyed.

The most recent August (moult) counts of harbour seals at haul-out sites in the UK and Ireland are presented for each country in Table 4. These data show an increasing trend for harbour seal counts within England and Wales, and a decreasing trend for Northern Ireland. There has been a generally decreasing trend for Scotland until the most recent survey period, where the data show a marked increase in harbour seal counts in 2013–2017. Overall, the UK total counts have increased since the last survey period and current numbers are at similar levels to counts from 1996–1997.

Table 4. The most recent August counts of harbour seals at haul-out sites in the UK by seal management unit, compared with three previous survey periods: 1996–1997, 2000–2006 & 2007–2009. Details of sources and dates of surveys used in each compiled regional total are given in SCOS (2018).

SEAL MANAGEMENT UNIT / COUNTRY	HARBOUR SEAL COUNTS			
	2013–2017	2007–2009	2000–2006	1996–1997
Scotland Total	26 553	20 430	23 391	29 514
England & Wales				
Total	5092	4035	3051	3160
Northern Ireland				
Total	948	1101	1176	*
UK Total	32 593	25 566	27 618	32 674*

*** No data available for Northern Ireland for 1996–1997.**

Grey seal

In the UK, 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. Results from the 2016 surveys, together with the 2017 estimates from the annually ground counted sites in eastern England and the most recent estimates for number of pups to have been born at less frequently surveyed colonies in Shetland, Wales, northwestern Scotland and Southwest England are presented in Table 2. These data provide a pup production estimate of 64 800 (rounded to the nearest 100) pups for the UK (no data are available for Northern Ireland). Trends of grey seal pup counts from subareas of the UK are shown in Figure 11.

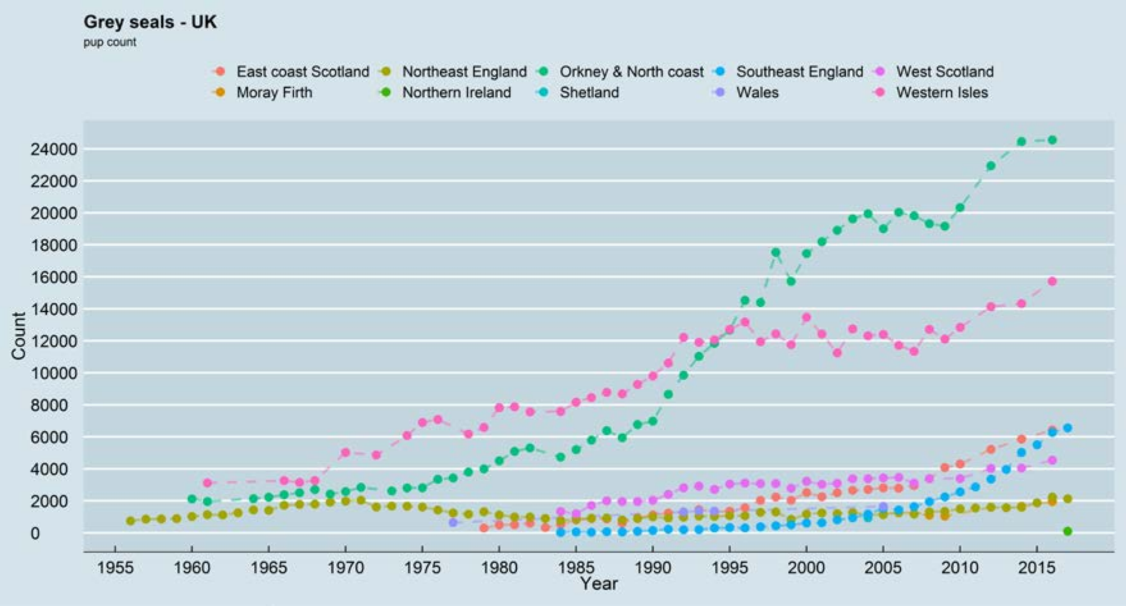


Figure 11. Trends of pup counts of grey seals in subareas of the UK.

1.1.1.6 North America

Harbour seals

In 2001, the harbour seal abundance in Maine was estimated at almost 100 000 individuals (Gilbert *et al.*, 2005). The growth of the harbour seal population along the US east coast is currently being reviewed, the results of this analysis should be available for the WGMME 2020 report.

Grey seals

Along the North American east coast, grey seal population trends are assessed from the counts of pups born during the breeding season. In 2016, the pup production on Sable Island accounted for 85% of the estimated total number of pups born in Canadian waters, with 11% in the Gulf of St Lawrence and 4% along the coast of Nova Scotia. The total population estimate in 2016 was lower than 2014, after adjustment for the sex ratio in the population and other changes to the population model (Hammill *et al.*, 2014; 2017). The total estimated Canadian grey seal population in 2016 was 424 300 (95% CI=263 600 to 578 300), with a Sable and Coastal Nova Scotia herd of 380 300 (95% CI=234 000 to 517 200), and 44 100 (95% CI=29 600 to 61 100) for Gulf of St Lawrence herd.

A smaller, but growing number of grey seal pups are born along the US east coast in Maine and Massachusetts. The number of pups born at US breeding colonies can be used to approximate the total size (pups and adults) of the grey seal population in US waters, based on the ratio of total best population size to pups in Canadian waters (4.3:1). Using this approach, the population estimate in US waters is 27 131 (CV=0.19, 95% CI: 18 768–39 221) animals in 2016 (Hayes *et al.*, 2017). There is uncertainty in this abundance level in the US because life-history parameters that influence the ratio of pups to total individuals in this portion of the population are unknown. It also does not reflect seasonal changes in stock abundance in the northeast region for a trans-boundary stock. For example, roughly 24 000 seals were observed in southeastern Massachusetts alone in 2015 (Pace *et al.*, 2019), and an estimated 28 000–40 000 grey seals were estimated in southeastern Massachusetts in 2015, using correction factors applied to seal counts visible in Google Earth imagery (Moxley *et al.*, 2017). Observed counts of grey seal pups from the North American east coast are shown in Figure 12. The grey seal pup counts from the US coast in 2008–

2014 do not include Seal Island, which is the 2nd largest breeding site, in theory a few hundred pups would have been missed.

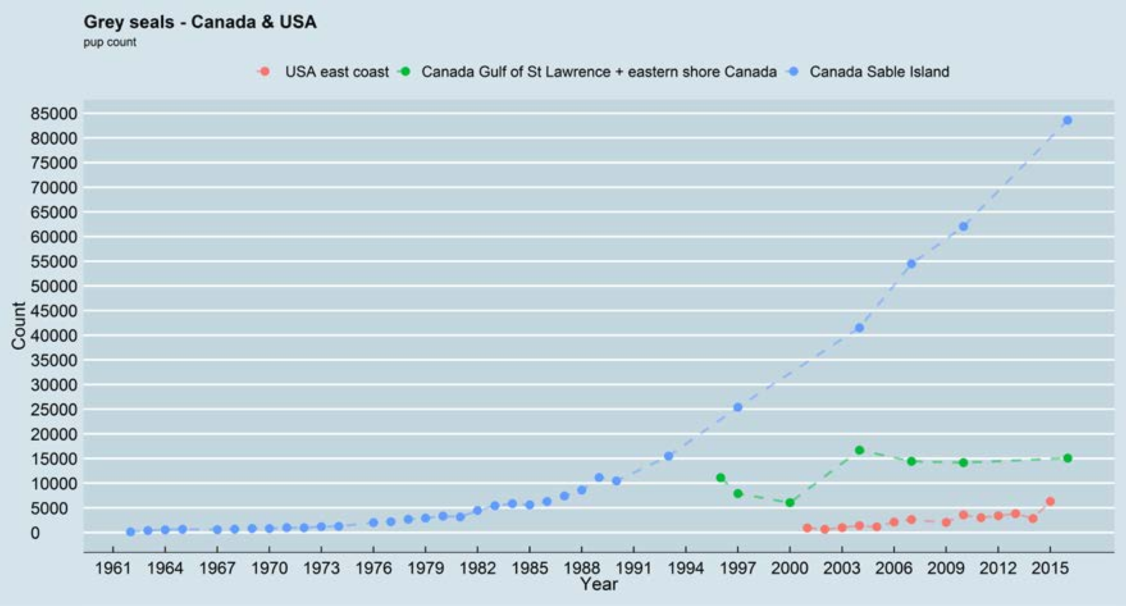


Figure 12. Trends of pup counts of grey seals along the east coast of North America.

1.1.2 Cetaceans

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

From May 2011 to April 2013, all EU Member States around the Baltic Sea carried out passive acoustic monitoring of harbour porpoises in the Baltic Sea as part of the SAMBAH project. Data were obtained at a total of 297 stations in waters of 5–80 m depth from the Darss and Drogden/Limhamn underwater ridges in the southwestern Baltic up to and including the Åland and Archipelago Sea in the north. Generalized additive models were used to describe the monthly probability of detecting porpoise clicks as a function of spatially referenced covariates and time. During May-Oct, two major clusters of harbour porpoise distribution were identified. The spatial segregation was most obvious during May–August, i.e. during the reproductive period. One cluster was centred on and around the shallow offshore banks southwest of the Island of Gotland, Sweden, proposed to represent the main part of the critically endangered Baltic Proper population. The other cluster had higher detection probabilities and was found west of the island of Bornholm, Denmark, proposed to consist of porpoises from the Belt Sea population, which is centred in the Danish Straits. Due to the spatial separation of the two clusters, a management border was proposed following a diagonal line, approximately between Hanö, Sweden, and Słupsk, Poland, during May-Oct. During Nov-Apr, and particularly during January–March, detections were more spread out and there was no longer any spatial separation between the two populations. The detection probabilities increased along the coasts of Poland, Lithuania, southern Latvia, along the Swedish east coast, and in southwestern Finnish waters. Also, during November–April, the detection probabilities were highest in the Danish waters in the southwestern part of the study area (Carlén *et al.*, 2018).

In 2018, WGMME reported on passive acoustic monitoring of harbour porpoises using C-PODs in Denmark, Finland, Germany, Poland and Sweden (ICES, 2018). The following new information can be added:

FINLAND: The Ministry of Environment and the Åland Government has granted funding to Turku University of Applied Sciences to continue the monitoring at least until autumn 2019. New data show the same seasonal pattern and similar detection rates as in previous years, indicating that harbour porpoises are present on a regular and predictable basis in the monitored area, however in small numbers. An update on the status of harbour porpoise in the northern and eastern Baltic Sea is planned to be published in near future (Loisa, pers. Comm).

GERMANY: With the aim of evaluating different monitoring methods, the German Oceanographic Museum has deployed an additional ten stations following an extended SAMBAH grid in the southwestern Baltic Sea from July to October 2018. In addition to C-PODs, four of the stations were equipped with Sound traps, giving a total of approximately seven weeks of hydrophone recordings (continuous sampling at 576 kHz). The study is planned to continue in 2019 with all ten stations equipped with both C-PODs and Sound traps, and a classifier will be used to extract harbour porpoise clicks from the Sound trap data. The acoustic data are to be compared to visual data collected by aerial surveys using both digital systems and observers collected in the same area and during the same time frame. The project is funded by the Federal Agency of Nature Conservation (Schwarzbach, pers. Comm.).

POLAND: The two-year monitoring programme (spring 2016 to spring 2018) supervised by Chief Inspectorate for Environmental Protection (CIEP) as part of the pilot monitoring (2015–2018) is completed. Monitoring was conducted in two areas (five stations within each area) where large numbers of positive detections of harbour porpoises were found during the SAMBAH project. At each monitoring station, porpoises were detected in every month of the year. In the Pomeranian monitoring site, the detection rate was twenty times higher than during the SAMBAH project, and in the Stilo monitoring site, it was four times higher.

The current monitoring scheme (under State Monitoring Programme) is planned to proceed with two continuous years of monitoring within six-year periods. The next two-year monitoring period is planned to start in 2022, subject to possible changes due to future HELCOM recommendations. There are three planned study areas: the same two as during the pilot monitoring, and a new one in the Gulf of Gdańsk, using the same stations as during SAMBAH (Kaminska, pers. Comm.).

Twenty-five C-PODs have been deployed in Puck Bay, a “hot spot” area for harbour porpoise in Polish waters, from November 2017 until October 2018. The acoustic monitoring resulted in a number of detections mostly in the outer part of the Bay. The study confirmed the year-round presence of harbour porpoise in the area, which has the most intensive gillnet fishery activity in Polish waters (Pawliczka, pers. Comm.).

SWEDEN: The monitoring carried out by the Swedish Museum of Natural History, funded by the Swedish Agency for Marine and Water Management, will continue at least until autumn 2019. Data on detection rates are uploaded on an annual basis to a publicly accessible database, Sharkweb, hosted by the Swedish Meteorological and Hydrological Institute. So far, the data for 2017–2018 have been uploaded.

The national monitoring data can be used for detecting changes in occurrence and/or distribution on local or possibly national scales. However, to estimate population abundance, and to investigate trends in population abundance and distribution patterns, population-wide surveys are needed. A SAMBAH II project is currently being designed. It aims to provide an updated abundance estimate with a narrower confidence interval, maps of harbour porpoise density (and not only probability of detection), and a further investigation of the spatial separation between the Baltic Proper and Belt Sea populations. The plan is to apply for funding from the EU Life programme in 2019. If successful, the project could start in summer/autumn 2020, and the passive acoustic data collection in summer/autumn 2021, i.e. approximately ten years after the initiation of the first SAMBAH project. The application process is being led by the Swedish Museum of Natural History.

Visual monitoring and strandings

BELGIUM: RBINS completed three aerial surveys in 2018. Densities in July and October were average, with on average 0.7 and 0.6 animals/km² respectively. The survey in April yielded a remarkably high average density (5.7 animals/km² in the survey area) with 404 animals sighted during the survey that lasted 3h44′ (on effort time). The animals were not evenly distributed, with very high densities (over 15 animals/km²) between the Westhinder anchorage area and the Norhthinder Traffic Separation System, a zone that is proposed as an offshore windfarm area (to be confirmed in the new marine spatial plan 2020–2026). (Preliminary data RBINS, unpublished).

In 2018, numerous bottlenose dolphins were observed in Belgian waters. On two occasions (April and October), groups comprising 25+ animals, including a number of calves, were observed. Other observations concerned solitary animals (at least two individuals, but possibly more), often very sociable towards humans, and staying in small areas for months.

The number of stranded harbour porpoises in 2018 (n = 89) was around the average of the last decade (Figure 13). The stranding pattern of porpoises (n= 1364, 1995–2017) has shifted seasonally: the peak was in March–May in the 20th century and has been in the period July–October in this century (Haelters *et al.*, 2018). The most remarkable stranding in 2018 was an 18 m long male fin whale of an estimated weight of 30 tons. The animal was freshly dead and had a thin blubber layer.

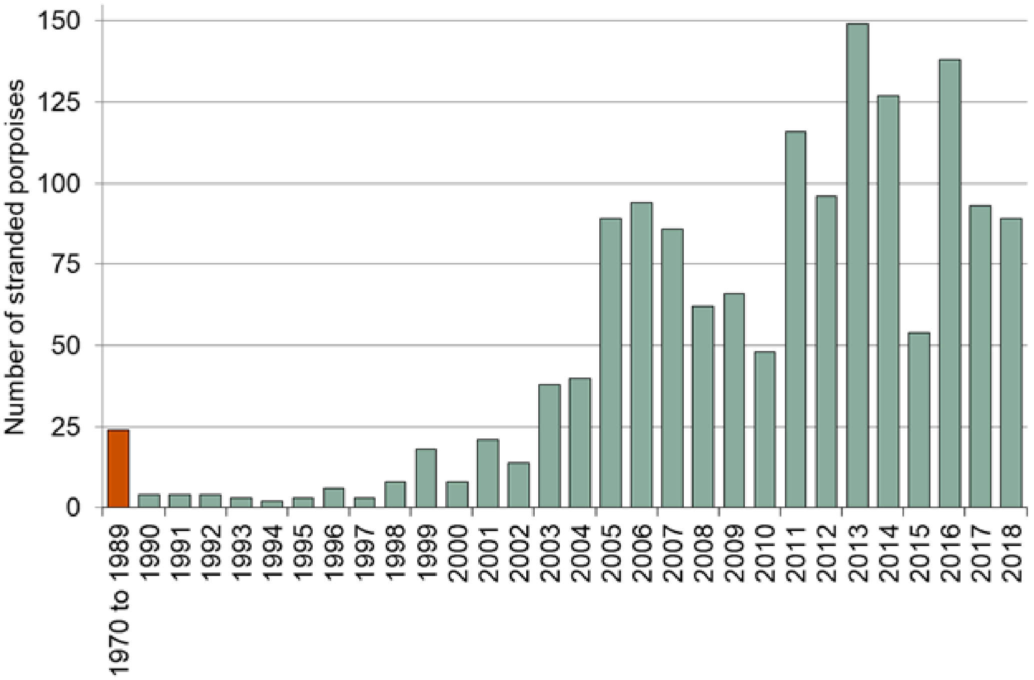


Figure 13. Strandings of harbour porpoises in Belgium recorded annually from 1990 to 2018 (plus total for 1970–1989). Data from Haelters *et al.*, (2018) and RBINS (unpubl.).

DENMARK: Kinze *et al.* (2018) presented an overview of cetacean strandings for the entire Danish coastline covering the period 2008–2017 and provided a comparative analysis with the preceding four decadal periods (1968–2007). Apart from harbour porpoises, 12 species totalling 89 individuals, were recorded during this period (Table 5). Harbour porpoise was the most numerous found species (n = 1177). The majority of the porpoises (62%) were found in the Outer Danish Waters (North Sea and Skagerrak). Smaller numbers (37%) were found in the Inner Danish Waters (Kattegat and Belt Seas), but the recorded numbers here are probably underestimated relative to the outer waters, due to the long coastline (ca 8000 km) along numerous islands and straits. Stranded animals were scarce in the western Baltic, 0.6% of the total. Numbers fluctuated yearly, without a detectable trend.

Table 5. Summary of cetacean strandings along the Danish coastline over a 50-year period from 1968 to 2017. Harbour porpoise is not included, since there was a break in recording effort in 2015–2017.

Species	1968–1977	1978–1987	1988–1997	1998–2007	2008–2017	Total
White-beaked dolphin	6	19	41	69	49	184
White-sided dolphin	1	0	2	6	3	11
Common dolphin	0	1	0	8	6	15
Striped dolphin	0	0	0	1	1	2
Bottlenose dolphin	4	0	0	0	0	4
Long-finned pilot whale	0	2	8	4	4	18
Killer whale	1	1	2	1	1	6
Risso's dolpin	0	0	0	0	1	1
Beluga whale	1	1	0	0	0	2
Sowerby's beaked whale	1	0	1	0	1	3
Northern bottlenose whale	1	1	1	0	0	3
Sperm whale	1	4	39	1	7	51
Minke whale	3	5	6	18	14	43
Bryde's whale	0	0	0	1	0	1
Sei whale	0	1	0	0	0	1
Fin whale	0	0	0	0	2	2
Humpback whale	0	0	0	0	1	1
Total	19	35	100	109	89	348

GERMANY: In 2017, the strata and transect design for the visual monitoring of harbour porpoises was revised in an effort to harmonise the national monitoring efforts for cetaceans and seabirds and to provide a survey design for potential future digital surveys. This resulted in the design of new study areas for the aerial line transect surveys in the German North Sea and Baltic Sea (Figure 14).

In spring 2017, one aerial line transect survey was conducted near Borkum Reef Ground and a total of 18 harbour porpoise groups (23 animals, incl. two calves) were sighted along 559 km of effort (Figure 15a). Due to logistical reasons and bad weather, no surveys could be conducted in the North and Baltic Sea during summer 2017.

In spring 2018, a total of 163 harbour porpoise groups (179 animals, no calves) were recorded along 1459 km of effort in three areas in the North Sea (Borkum Reef Ground, Weser-Elbe estuary and Dogger Bank). In summer 2018, a total of 166 groups (200 animals, incl. 14 calves) were observed under 2077 km of effort in four study areas in the North Sea (Weser-Elbe estuary, Sylt Outer Reef West and East, and Dogger Bank, Figure 15b). Three sightings of minke whale (single animals) were observed near the Dogger Bank (two in spring, one in summer).

The Baltic Sea was surveyed in five study areas (Kiel Bight, Fehmarn, Mecklenburg Bay West and East, North of Ruegen) in summer 2018 and a total of nine harbour porpoise groups (ten animals, incl. one calf) were sighted along 1154 km of effort (Figure 15c). This limited number of sightings in the Baltic Sea may have been due to unfavourable weather conditions during the surveys (sea state ≤2 in 70% of the time).

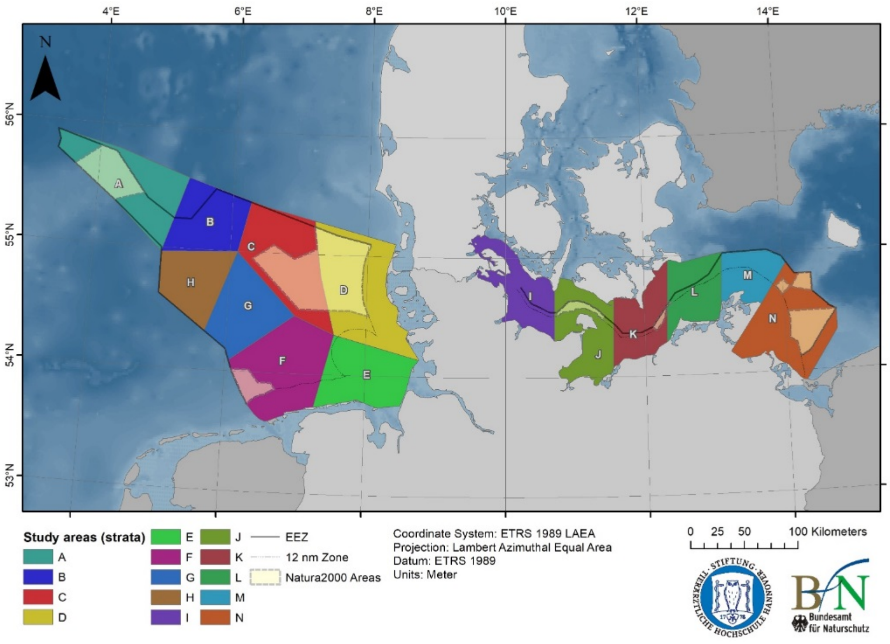
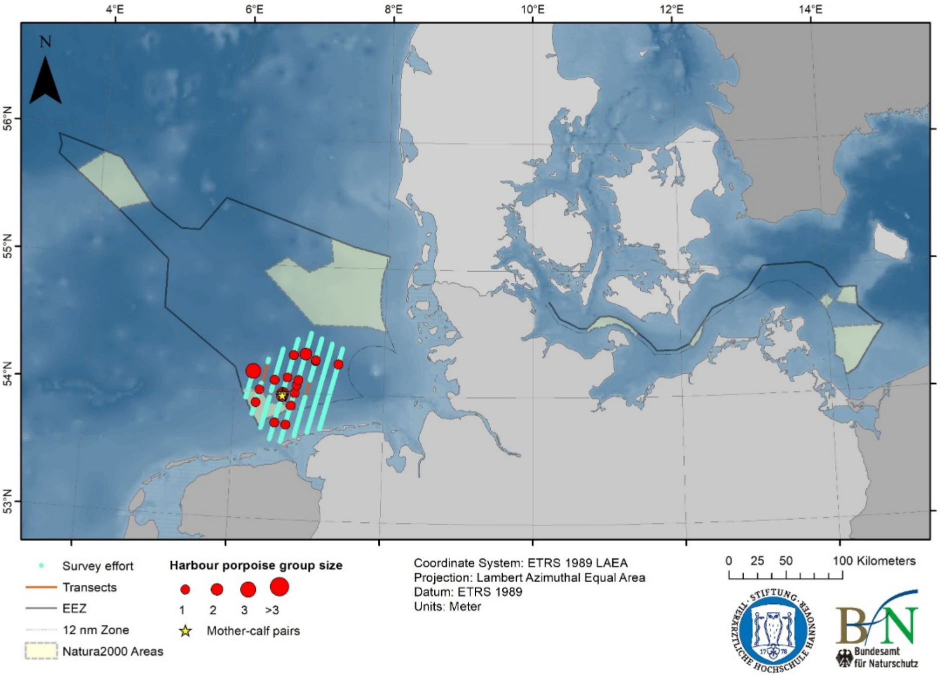


Figure 14. Newly designated study areas for the visual monitoring of harbour porpoises in the German North and Baltic Sea.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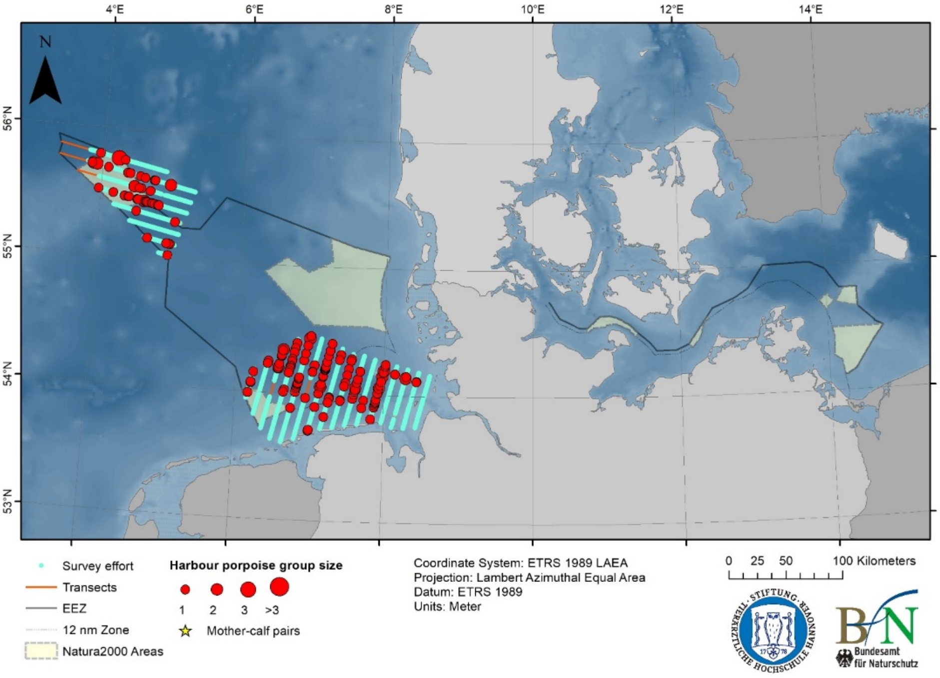
Bundesamt für Naturschutz

a)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Harbour porpoise group size
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Mother-calf pairs
Coordinate System: ETRS 1989 LAEA
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Datum: ETRS 1989
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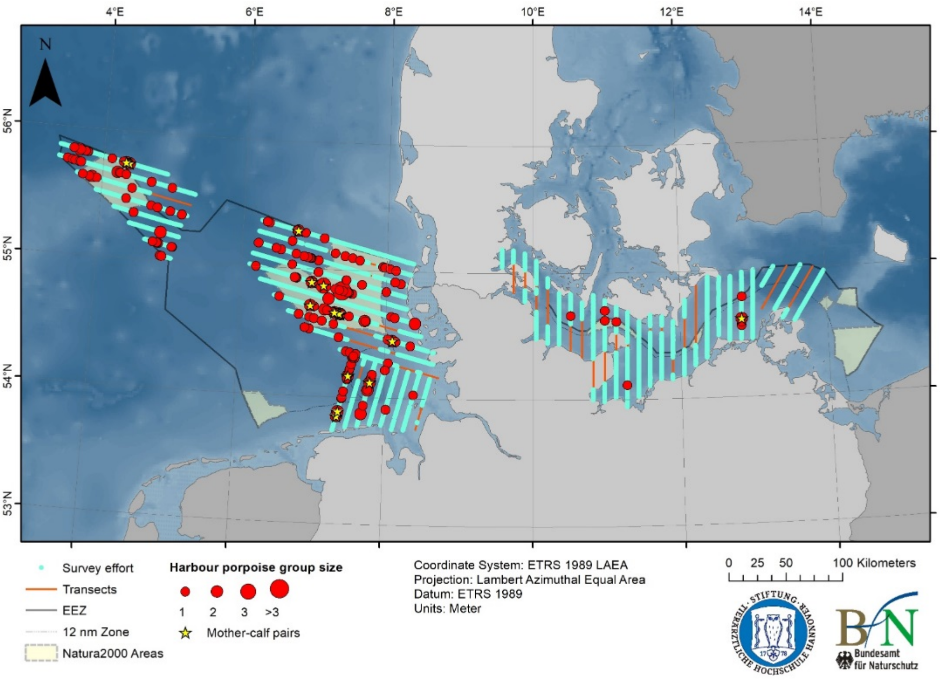
Bundesamt für Naturschutz

b)



c)

Figure 15. Survey effort and harbour porpoise sightings during aerial surveys in the German North and Baltic Sea during a) spring 2017, b) spring 2018 and c) summer 2018. Harbour porpoise group sizes are indicated using group size dependent red circles; yellow stars mark mother-calf pairs; red lines indicate transect lines that were not covered though planned; blue lines indicate covered transect lines (i.e. survey effort).

Effort corrected density and abundance estimates were generated using a bootstrapping approach, also correcting for availability and perception bias. In spring 2017, the abundance for Borkum Reef Ground in the North Sea was estimated to be 2862 (95%CI: 1175–4656) animals, at 0.44 (0.19–0.76) animals/km². In spring and summer 2018, the German North Sea was not entirely covered, allowing abundance and density estimates only for the individual areas (Table 6). The German Baltic Sea area (areas I-M), however, was largely covered, leading to a total abundance estimate of 874 (265–1765) harbour porpoises and an average density of 0.06 (0.01–0.12) animals/km² (Table 6).

Table 6. Summary of effort corrected, bootstrapped density and abundance estimates for spring 2017, and spring and summer 2018 in the study area of the North and Baltic Sea (see Figure 14 for areas). N = estimated abundance of harbour porpoises; N95%CI = 95% confidence interval around N; D = density estimate of harbour porpoises (animals/km²); D95%CI=95% CI around D; s = average group size.

AREA	SEASON	N	N95% CI	D	D95% CI	Ŝ
Borkum Reef Ground (F)	spring 2017	3571	1330–7348	0.47	0.22–1.21	1.25
Dogger Bank (A)	spring 2018	5192	2287–8759	0.92	0.41–1.55	1.15
Weser-Elbe estuary (E)	spring 2018	3298	1391–6285	0.75	0.32–1.44	1.12
Borkum Reef Ground (F)	spring 2018	5808	3175–9123	0.95	0.52–1.50	1.05
Dogger Bank (A)	spring 2018	5192	2287–8759	0.92	0.41–1.55	1.15
Dogger Bank (A)	summer 2018	4175	2570–6228	0.74	0.46–1.10	1.13
Sylt Outer Reef West (C)	summer 2018	4937	2694–8029	0.82	0.45–1.34	1.18
Sylt Outer Reef East (D)	summer 2018	5262	2306–9098	0.76	0.33–1.32	1.35
Weser-Elbe estuary (E)	summer 2018	3410	383–7064	0.78	0.09–1.61	1.20
Kiel Bight (I)	summer 2018	182	0–653	0.06	0–0,21	1.00
Fehmarn (J)	summer 2018	388	115–677	0.11	0.03–0.18	1.00
Mecklenburger Bay West (K)	summer 2018	0	0–0	0	0–0	
Mecklenburger Bay East (L)	summer 2018	304	0–1009	0.10	0–0.34	1.25
North of Ruegen (M)	summer 2018	0	0–0	0	0–0	

FRANCE/Surveys: The recurrent cetacean and seabird sighting programmes conducted on board RV Thalassa during the fish stock assessment surveys PELGAS, IBTS, CGFS and EVHOE continued during 2018.

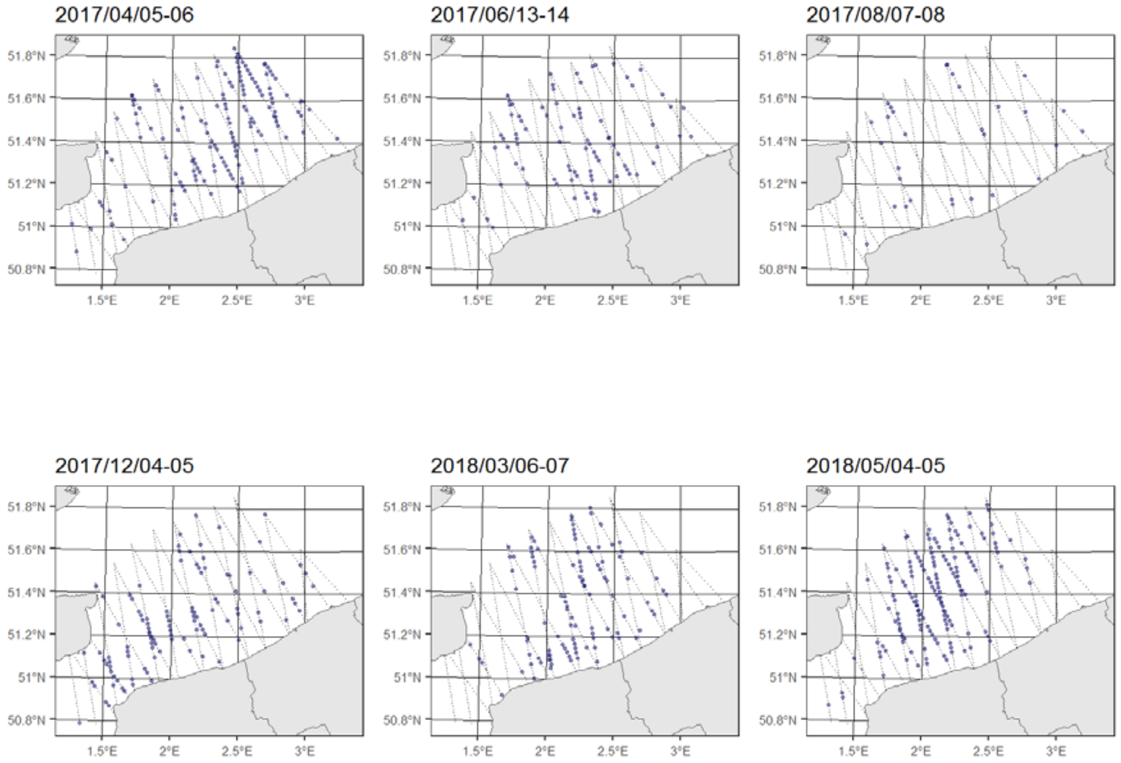
The integrated ecosystemic PELGAS ("Pélagiques Gascogne") survey carried out every year during spring in the Bay of Biscay was carried out at the community level over more than a decade (2004–2016) (Authier *et al.*, 2018). Lambert *et al.* (in press) investigated the functional prey–predator links between two predator species: short-beaked common dolphins and bottlenose dolphins with a range of prey species (small forage fish, e.g. clupeids) in the Bay of Biscay. The study showed a negative spatial relationship between estimated prey biomass and predator sightings that may reflect prey avoidance.

In 2017–2018, a specific survey was dedicated to estimate marine mammal and seabird relative abundance and distribution in the area of Dunkirk (northern France) before construction of an offshore windfarm (Virgili *et al.*, 2018). The survey effort covered 9400 km^2 distributed as follows: 37% in France, 37% in Belgium and 26% in UK. Observations were collected following a standardised aerial survey protocol (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). In 2018, two sessions were realised on 6–7 March (1256 km) and 4–5 May (1526 km).

The most sighted marine mammal species was the harbour porpoise and the number of observations reflected a high seasonality for this species (Table 7). Harbour porpoise distribution also differed between the sessions (Figure 16).

Table 7. Observations of harbour porpoises during the aerial survey (number of detections on effort; Virgili *et al.*, 2018)

	April 2017	June 2017	August 2017	December 2017	March 2018	May 2018
Harbour porpoise	315	100	35	202	147	321



2017/04/05-06

2017/06/13-14

2017/08/07-08

2017/12/04-05

2018/03/06-07

2018/05/04-05

Figure 16. Observations of harbour porpoises in the eastern Channel during 2017–2018. Dotted lines are the transect lines, and blue dots are the detections of harbour porpoises.

FRANCE/Strandings: Strandings have been recorded since the early 1970s in France by the National Stranding Network (Réseau National d'Echouage, RNE). Since the network is considered to be relatively dense and stable since the 1980s and the reporting has been stable since the early 1990s, it is assumed that observed fluctuations or trends since 1990 reflect biological or physical parameters such as abundance, mortality or drift conditions. The overall number of cetacean strandings in 2017 (n = 1642) is the highest ever, and well above the average of the last ten years estimated at 890 strandings per year (red line Figure 17). The time-series shows an overall increase of strandings along French coasts (Dars *et al.*, 2018).

Twelve cetacean species were reported in 2017 (Figure 18), showing a species composition comparable to the previous years, with common dolphin as the most represented species, (67.2% of the strandings), followed by harbour porpoise (19.6%) and striped dolphin (8.2%).

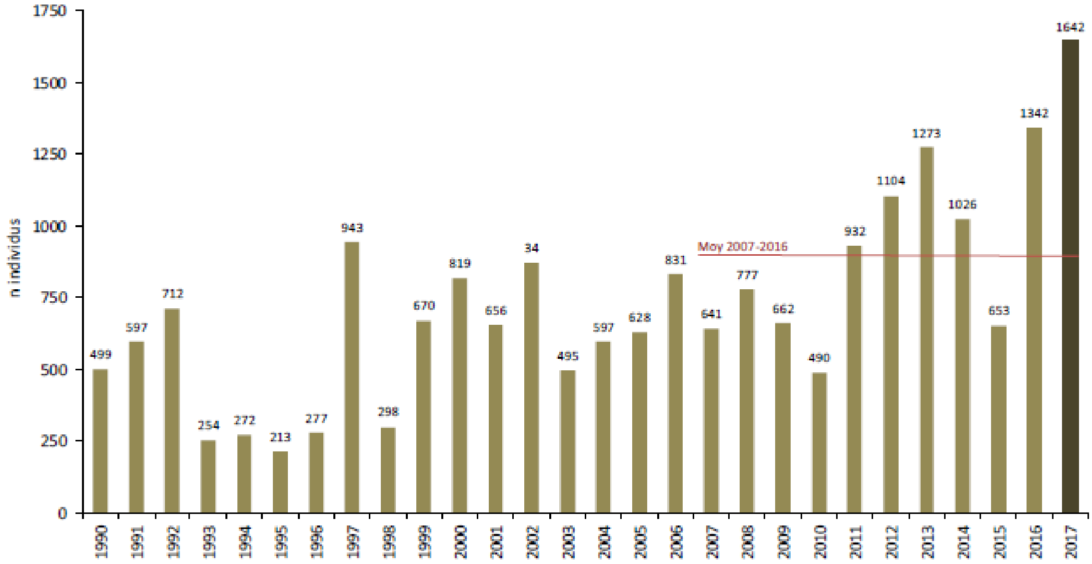


Figure 17. Annual distribution of the number of cetacean strandings along the French coasts from 1990 to 2017 (n = 20 175) (Dars *et al.*, 2018).

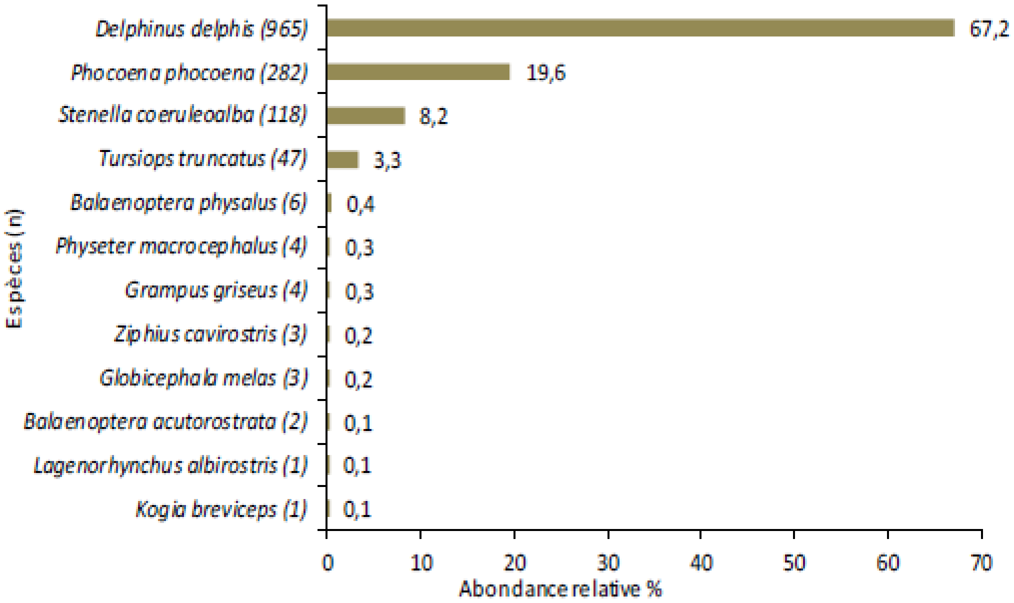


Figure 18. Relative abundance of the cetacean species stranded along the French coasts in 2017 (n = 1436; 206 unidentified individuals were excluded. (Source: Dars *et al.*, 2018).

IRELAND: WGMME (ICES, 2018) reported on the aims and methods of the ObSERVE Programme, consisting of acoustic and aerial surveys in 2015 and 2016. The final reports of this programme were published recently (Berrow *et al.*, 2018; Rogan *et al.*, 2018a).

The acoustic survey recorded 13 species, including five mysticetes and eight odontocetes (Berrow *et al.*, 2018). Blue whale infrasonic moans were recorded in summer and autumn months. The number of blue whale calls per day varied significantly according to month, season and acoustic monitoring station. Fin whale detections occurred at all acoustic monitoring stations and in all seasons, with mean detection counts per hour lowest in summer and highest in autumn. Detection rates were highest at northern stations and varied significantly according to the month, season and acoustic monitoring station, with significantly more detections recorded at night. Humpback whales were rarely detected, with most detections from the western and south-western zones. Sperm whale clicks were detected at all acoustic monitoring stations in all seasons, but detections were lowest at the two most southerly stations in all seasons. The number of sperm whale clicks per day varied significantly according to month and season, with a northerly movement from spring to autumn, and with more detections made at night. Sperm whales occurred over the Erris Basin in all seasons, with more detections on PAM over the Porcupine Shelf in summer and off the Goban Spur in autumn. Sowerby's beaked whale clicks were recorded at all acoustic monitoring stations in all seasons, with the highest rate recorded at the most northerly station in spring. With the exception of the most northerly station in spring and autumn, Cuvier's beaked whales were detected at all acoustic monitoring stations in all seasons. The highest detection count per hour for Cuvier's occurred in the spring at southern stations, which was the opposite pattern of Sowerby's beaked whale. Northern bottlenose, minke and sei whales were only occasionally detected during the two-year field study.

The identification of dolphins to species level using clicks was not feasible. Harbour porpoises were detected during spring, summer and autumn PAM surveys in 2015. Clicks were recorded in waters surrounding the shelf edge, with one detection occurring over deeper water in the Rockall Trough.

Density estimates could be generated for sperm whales using distance sampling as target motion analysis was possible for sperm whale clicks. Density estimates ranged from one to 4.6 individuals per 1000 km^2 in different areas. Density estimates will always be higher when restricted to areas of favourable habitat, such as offshore from the shelf break, but the surveys were designed to provide good coverage, both inshore and offshore of the continental slope, thus covering areas where lower densities were expected. No sperm whales were detected in waters less than 300 m deep and the overall density for the whole surveyed region excluding waters less than 300 m was 3.2 individuals per 1000 km^2. An overall cumulative abundance estimate in the entire study area was 380 individuals.

Rogan *et al.* (2018a) reported on the results of the ObSERVE aerial surveys. Over the two-year survey period, there were 1844 sightings of at least 19 cetacean species, comprising an estimated 8633 individuals during 10 255 and 10 143 km on effort in summer and winter respectively. In both years, more cetacean sightings occurred in the winter period than in the summer period. Species richness was also higher in winter than in the summer periods. Bottlenose dolphins, harbour porpoise and common/striped dolphins were the most frequently sighted odontocetes, whereas minke whale was the most frequently sighted mysticete species.

Using habitat modelling, abundance estimates have been calculated for both summer and winter. The study highlights that there are both significant seasonal and interannual differences in abundance for a number of cetacean species, although the drivers of species' habitat use, and the distributions observed in 2015 and 2016 are not well understood. Densities of short-beaked common dolphins and bottlenose dolphins were much higher during winter than summer, whereas harbour porpoises and minke whales were more commonly sighted in summer. Design-based and/or model-based estimation methods were used to estimate abundance for a range of species. Abundance estimates for smaller dolphin species, including the short-beaked common dolphin, striped dolphin and common bottlenose dolphin, were 80 763 individuals in summer (CV = 0.15) and 145 173 in winter (CV = 0.10), long-finned pilot whales: 6235 individuals in summer (CV = 0.29) and 6103 in winter (CV = 0.26); harbour porpoise: 35 975 individuals in summer (CV = 0.09) and 20 571 in winter (CV = 0.23), and minke whale: 8311 individuals in summer (CV = 0.23) and 3873 (CV = 0.19) in winter. Best estimates of abundance for all beaked whales using seasonal data from both survey years were 2327 individuals in summer (CV = 0.39) and 3997 (CV = 0.38) in winter.

There were clear changes in observed distribution on a seasonal basis for a number of species. Here, we present the density surface models with the data aggregated on a seasonal basis (both years combined) to illustrate seasonal use (Figure 19). For harbour porpoise, the summer predicted distribution suggests that this species uses large areas of the continental shelf, including the Celtic Sea and are mostly, but not exclusively found in waters <200 m depth, although some sightings are of individuals seen in much deeper waters, including a sighting in waters >2000 m. In contrast, the winter distribution model suggests that shallower waters and in particular the coastal waters are more important for the species during winter, suggesting some changes in habitat use seasonally. In both seasons, the Irish Sea emerged as an important area for harbour porpoises.

In contrast, predicted distribution maps of bottlenose dolphin suggest that the inshore areas southwest of Ireland and parts of the Porcupine Seabight are important to this species in summer. Bottlenose dolphin distribution was more widespread in winter, with the predicted habitat use suggesting a more "offshore" distribution, with areas around the 200 m contour important, as well as areas in much deeper waters.

Predicted summer distribution of the common dolphin and common dolphin/striped dolphin group was predominantly to the southwest of Ireland, and also in deeper waters off the edge of the continental shelf. In contrast, predicted winter distribution was more widely distributed, showing areas on the continental shelf and coastal waters to be important, including the southeast Celtic Sea and the northwest off Co Donegal. It seems that in both these years combined, there may be a north–south movement, with a more northerly distribution in winter.

Minke whale also appears to undergo seasonal movements with predicted distribution in summer showing a wide spatial distribution, including coastal and continental habitat use, whereas winter distribution was predicted to occur south and west of Ireland, with very little of the coastal area predicted to be of high use, including the Irish Sea.

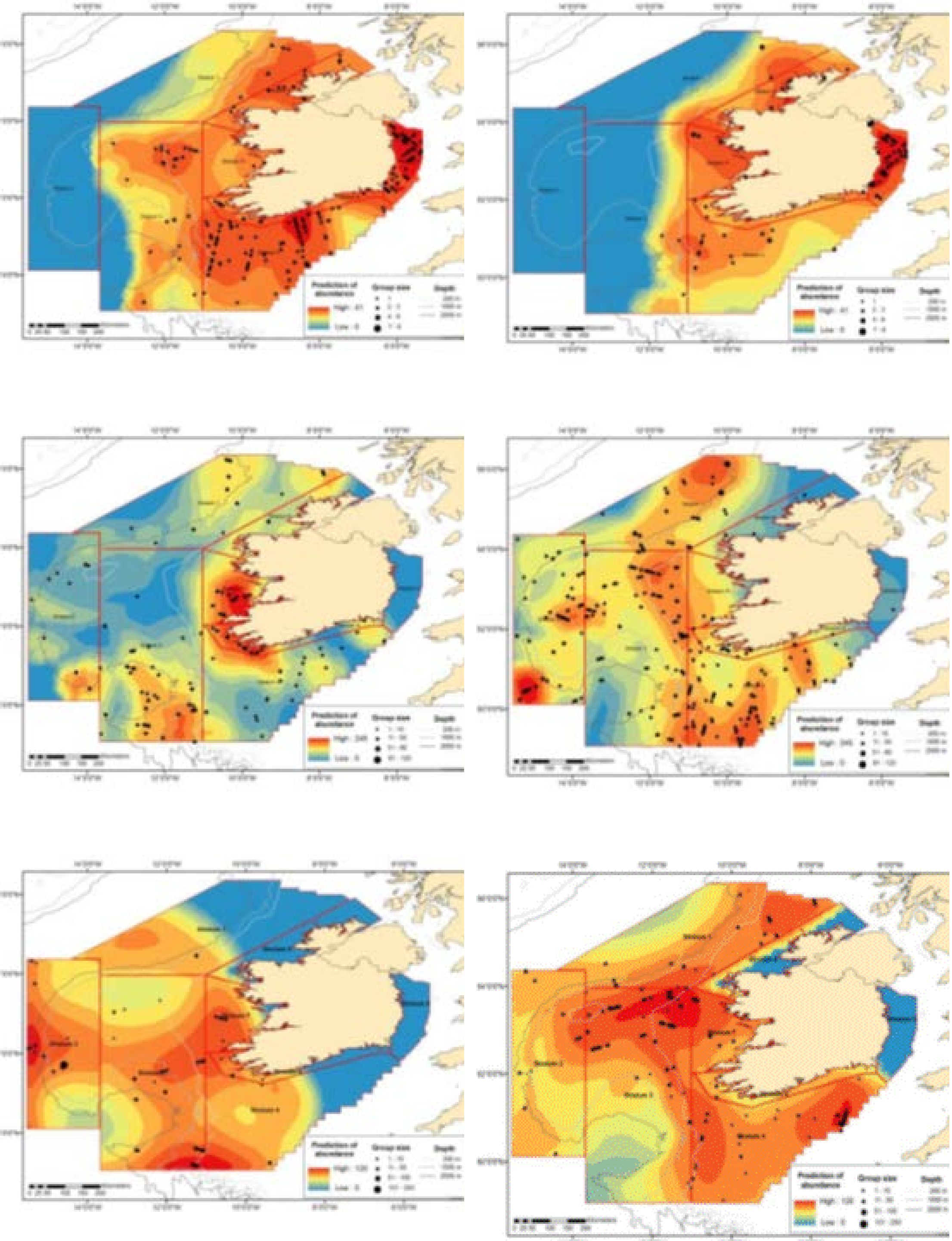


Figure 19. Predicted summer (left panel) and winter (right panel) distribution of harbour porpoise (top), bottlenose dolphin (middle), common+common/striped dolphin (bottom). Note that scale on density estimates is a relative figure, which varies for each species and does not represent the absolute abundance of animals.

Apart from the large-scale ObSERVE surveys, standardised boat surveys and photo identification of individual bottlenose dolphins are used to estimate the abundance of bottlenose dolphins in smaller coastal areas. These survey methods were used to make an abundance estimate of bottlenose dolphins using the Lower River Shannon SAC during summer and autumn of 2018. A total of 12 surveys were conducted between June and October resulting in 42 encounters with bottlenose dolphin schools. Using standard mark recapture methodology, the total number of dolphins using the Shannon SAC during June to early October 2018 was estimated as 139 dolphins (CV = 0.109, 95% confidence interval: 121–160). This estimate lies within the range of previous estimates calculated since 1997 indicating a stable population size (NPWS unpublished data, Rogan *et al.*, 2018b).

NETHERLANDS: In July 2018, aerial surveys to estimate the abundance of harbour porpoise on the Dutch Continental Shelf were conducted (Geelhoed *et al.*, 2018b). These surveys followed predetermined track lines in four areas: A - Dogger Bank, B - Offshore, C - Frisian Front and D - Delta. Between 13–18 July 2018, the entire Dutch Continental Shelf was surveyed, resulting in a total distance of 3039.8 km of effort. Of this effort, 79.5% was surveyed with good or moderate conditions on at least one side of the plane.

Harbour porpoises were assessed using line transect distance sampling methods. Density and abundance estimates were calculated. Porpoise densities varied between 0.54–1.76 animals/km² in the areas A–D. The overall density on the entire Dutch Continental Shelf was 1.07 animals/km² (Table 8). The distribution of harbour porpoises is shown in Figure 20.

The total number of harbour porpoises on the Dutch Continental Shelf (areas A–D) was estimated at 63 514 animals (CI = 34 276–119 734) in July 2018 (Table 8). The abundance estimate for the DCS in 2018 of 63 514 individuals (CI = 34 276–119 734) lies between the minimum estimate in 2010 (n = 25 998; CI = 13 988–53 623) and the maximum estimate in 2014 (n = 76 773; CI = 43 414–154 265). Neither the DCS abundance estimate, nor the abundance estimates per subarea show a trend (Geelhoed and Scheidat, 2018).

Table 8. Density and abundance estimate of harbour porpoise on the Dutch Continental Shelf in July 2018 per area.

	Density (animals/km²)	95% CI	Abundance (n animals)	95% CI	CV
Area A – Dogger Bank	0.63	0.30–1.43	6020	2859–13 704	0.41
Area B – Offshore	1.76	0.87–3.44	29 722	14 663–58 170	0.36
Area C – Frisian Front	1.38	0.63–2.84	16 595	7618–34 120	0.37
Area D – Delta	0.54	0.26–1.06	11 176	5400–22 078	0.35
Total DCS	1.07	0.58–2.02	63 514	34 276–119 734	0.32

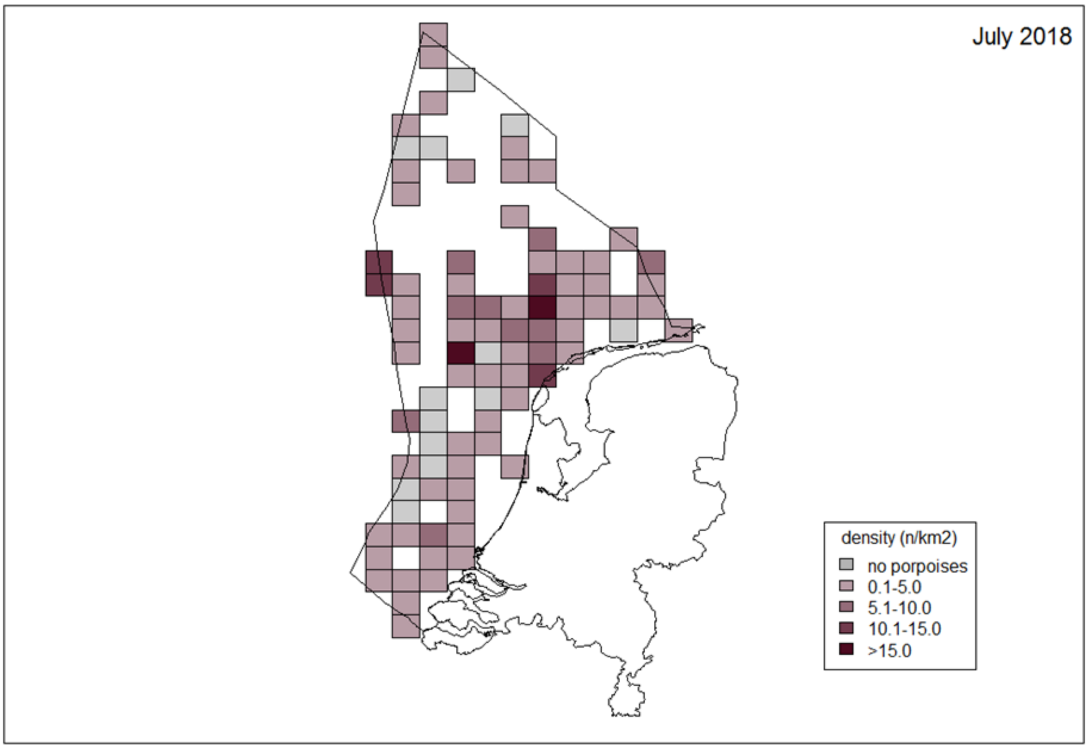


Figure 20. Density distribution of harbour porpoises (animals/km²) per 1/9 ICES grid cell, July 2018. Grid cells with low effort (<1 km²) are omitted.

In 2018, 471 stranded cetaceans divided over four species were recorded by Naturalis Biodiversity Center (www.walvisstrandingen.nl). As usual, harbour porpoise was the most abundant species (n = 468, Table 9). The number of stranded harbour porpoises in 2018 is among the lowest in the last decade, but is well above the average in years before that (Figure 21). Since 2016, ca. 50 fresh harbour porpoises are collected annually 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 (IJsseldijk *et al.*, 2018a). The 2018 results are not available yet.

Table 9. Stranded cetaceans recorded in the Netherlands in 2018. Source: www.walvisstrandingen.nl Naturalis Biodiversity Center.

Species	n
Harbour porpoise	468
Sperm whale	1
Long-finned pilot whale	1
Common dolphin	1
	471

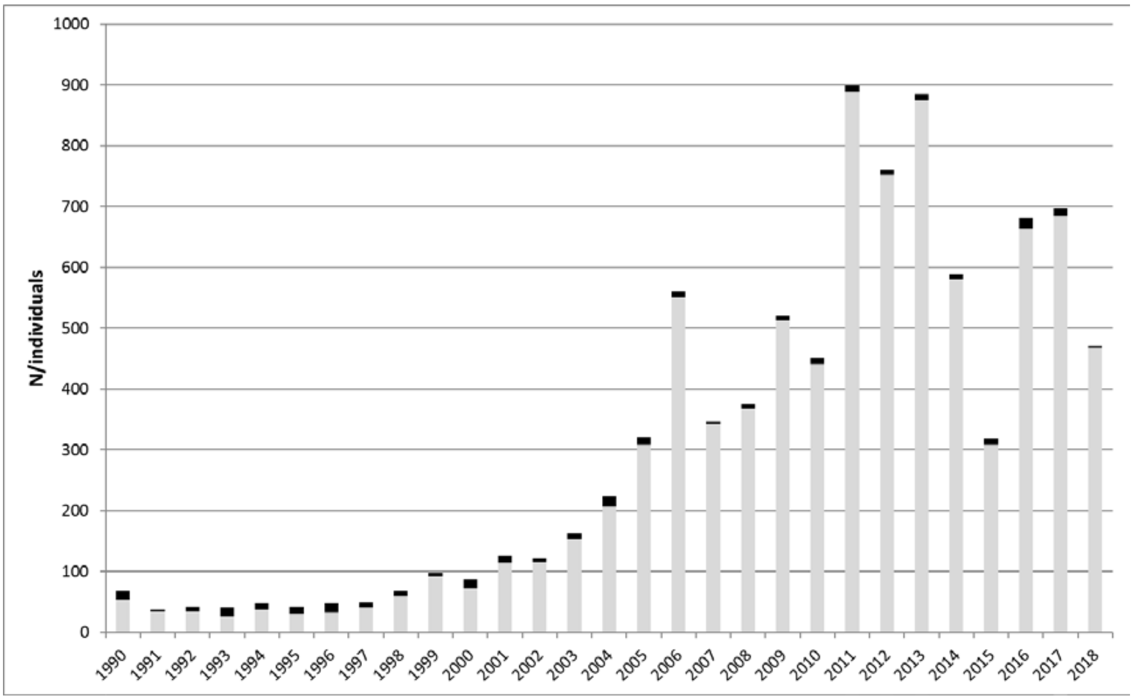


Figure 21. Strandings of harbour porpoises (grey) and other cetaceans (black) in the Netherlands recorded annually from 1990 to Source: www.walvisstrandingen.nl Naturalis Biodiversity Center.

POLAND: Strandings of harbour porpoises were continuously collected by Professor Krzysztof Skóra Hel Marine Station, University of Gdańsk in 2018. In cooperation with WWF Poland within the external project "Blue Patrol", 14 dead harbour porpoises were collected. They were too decomposed to perform a post-mortem analysis. The marks of human intervention (fins cut off, dorsal part cut) were observed in three of them. Fishermen operating in the central part of the Polish coast reported one bycaught animal in gillnets.

PORTUGAL: Systematic annual aerial surveys of cetaceans along the coast of mainland Portugal started in 2010 with the SafeSea project (discussed in ICES, 2014) and continued annually until 2015 in the LIFE+ MarPro project (Vingada and Eira, 2018). The LIFE+ MarPro aimed to define Natura 2000 sites for cetaceans and seabirds in mainland Portugal. Apart from coastal observations and a ship-based survey of offshore waters (50–200 nm out) in 2010, two types of aerial surveys were conducted; one aimed at seabirds and cetaceans extending up to the 50 nm (62 716 km^2), the other aimed at harbour porpoises in the 'near shore' area, restricted to the 200 meter bathymetric line (25 053 km^2, Figure 22). Density and abundance estimates were calculated using distance sampling methods.

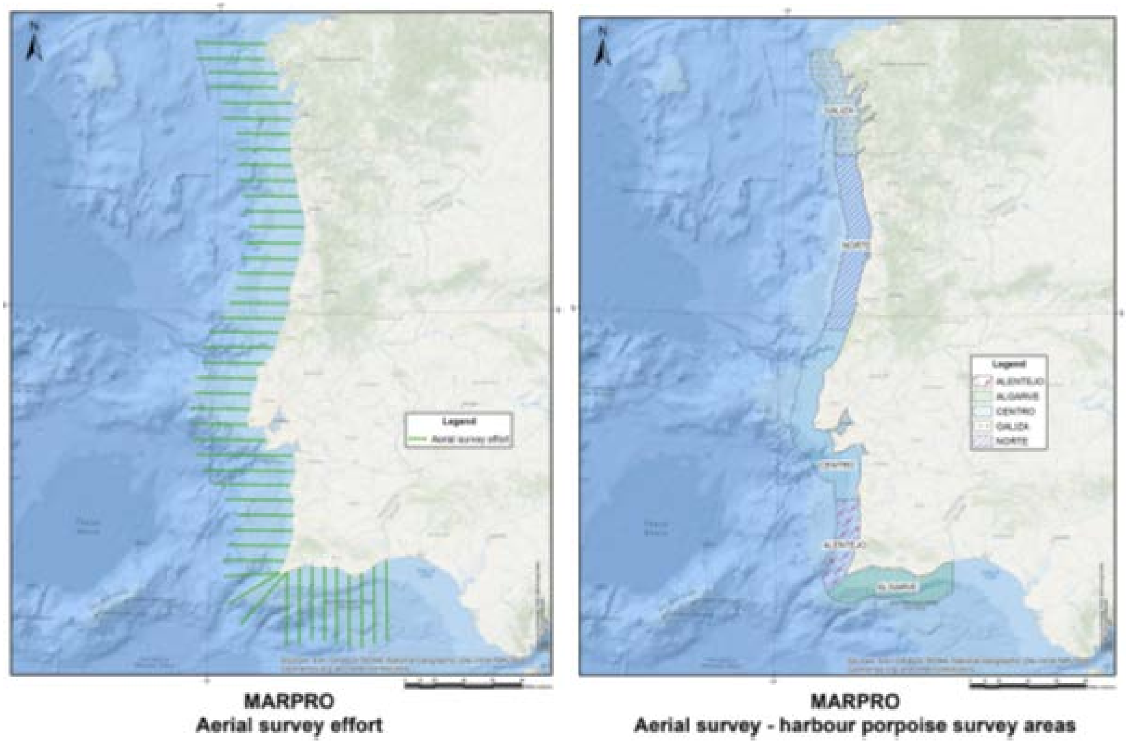


Figure 22. Design of aerial survey track lines. Left: all species except Harbour porpoise. Right: Harbour porpoise. From Santos *et al.*, 2012.

During the project, 28 species of cetaceans were recorded in the Portuguese mainland coast. It was possible to calculate abundance estimates for a few species (Table 10). Common dolphin and striped dolphin were the most abundant species, with total numbers of 45 179 (CV = 0.24) and 19 473 (CV = 0.44), respectively. Harbour porpoise and bottlenose dolphin were important target cetacean species as they are listed under Annex II of the Habitats Directive.

Table 10. Abundance estimates for the most abundant species in Portuguese mainland waters (2010–2015).

Species	Abundance (n)	CV
Harbour porpoise	1531	0.31
Bottlenose dolphin	2306	0.35
Common dolphin	45 179	0.24
Striped dolphin	19 473	0.44
Minke whale	1406	0.48
Fin whale	627	0.48

Harbour porpoise densities varied between 0.006–0.096 animals/km^2 in the years 2010–2015. The overall density in Portuguese near shore waters was 0.061 animals/km^2 (Table 11). These densities corresponded to an abundance estimate of Harbour porpoises in Portuguese mainland waters of 1531 (CV = 0.31) for the period 2010–2015, ranging from 153 (CV = 1.01) in 2010, to 3020 (CV = 0.37) in 2013 (Table 11). The species was distributed along the entire coast with the largest numbers north of Nazaré (Figure 23).

Table 11. Abundance estimates and densities of Harbour porpoise in Portuguese mainland waters (2010–2015).

Year	Density (animals/km^2)	Abundance (n animals)	CV
2010	0.006	156	1.01
2011	0.039	991	0.49
2012	0.096	2416	0.30
2013	0.077	3020	0.37
2014	0.059	1493	0.41
2015	0.074	1859	0.40
2010–2015	0.061	1531	0.31

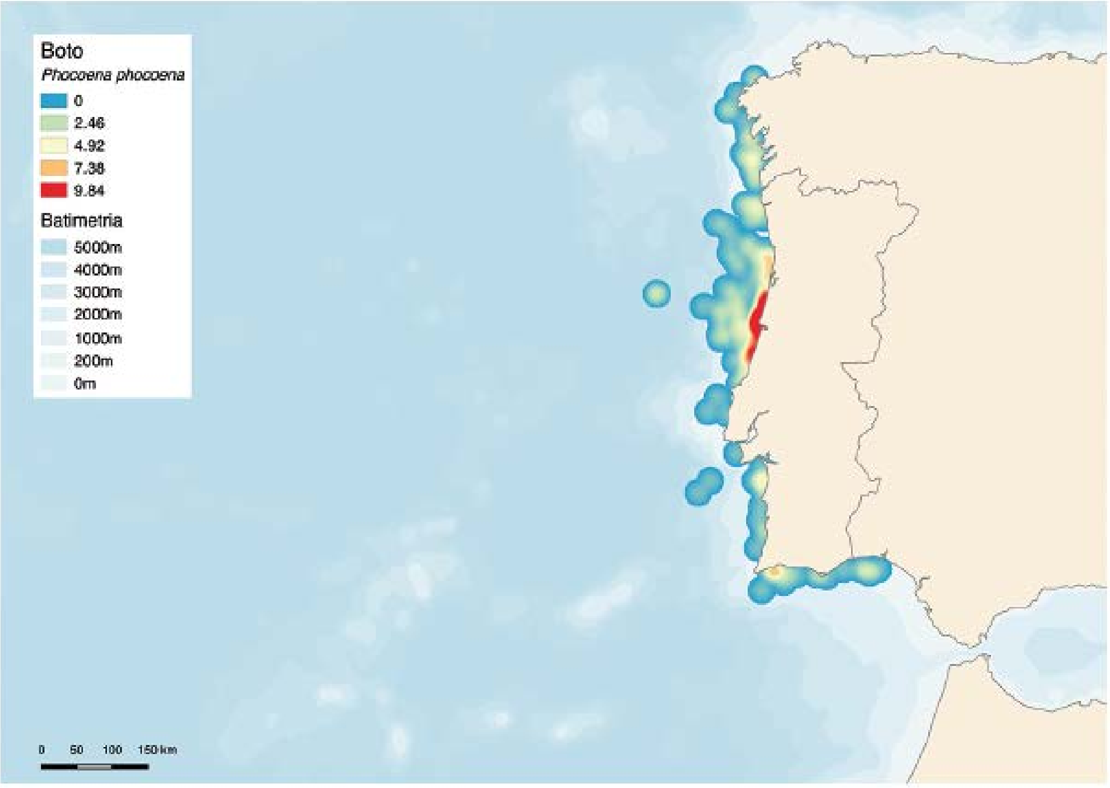


Figure 23. Harbour porpoise average abundance 2010–2015 (n= 1531, CV = 0.31). The map is based on Kernel density estimates showing zones with greater occurrence of porpoises.

Densities of bottlenose dolphins varied between 0.012–0.089 animals/km^2 in the years 2010–2015. The overall density in Portuguese near shore waters was 0.037 animals/km^2 (Table 12). These densities corresponded to an abundance estimate of 2306 (CV = 0.35) for the period 2010–2015, ranging from 736 (CV = 0.60) in 2010, to 5615 (CV = 0.61) in 2013 (Table 12). The distribution shows a continuous presence along the coast, with larger numbers north of Cabo Raso, in the area of Peniche-Nazaré, and in the area between Figueira da Foz and Porto (Figure 24).

Table 12. Abundance estimates and densities of bottlenose dolphin in Portuguese mainland waters (2010–2015).

Year	Density (animals/km^2)	Abundance (n animals)	CV
2010	0.012	736	0.60
2011	0.040	2521	0.61
2012	0.017	1063	0.88
2013	0.089	5615	0.61
2014	0.056	3529	0.92
2015	0.020	1252	0.43
2010–2015	0.037	2306	0.35

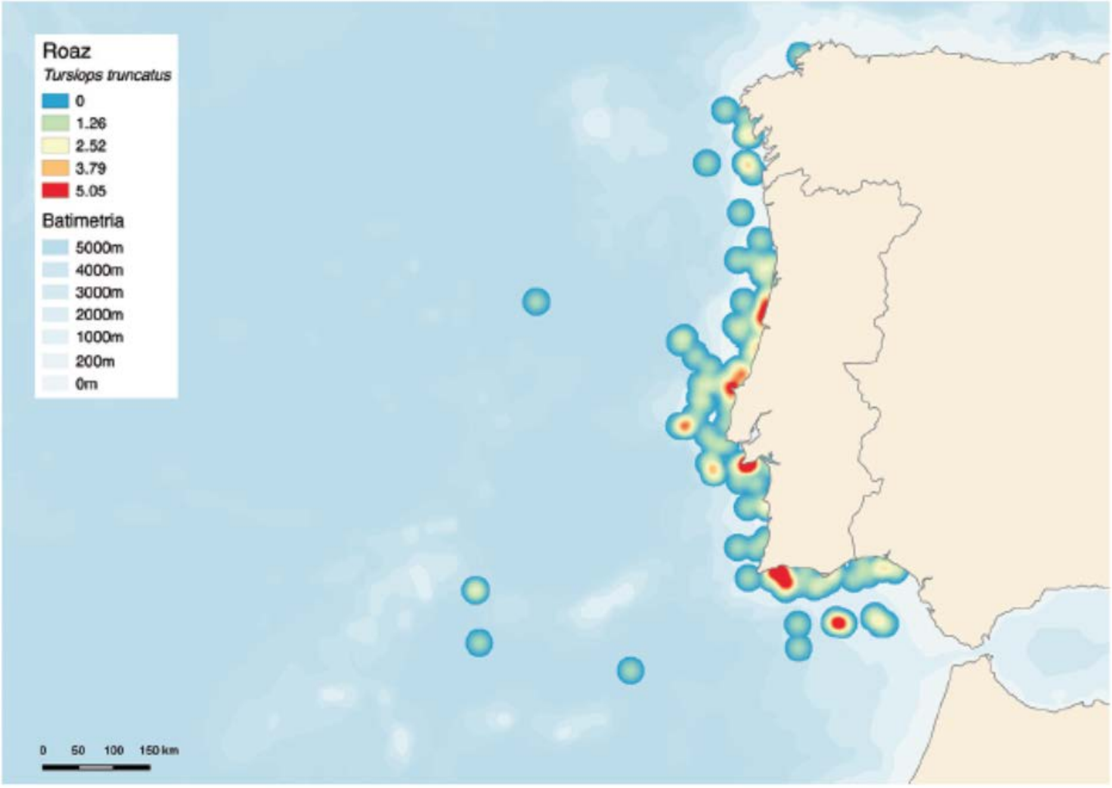


Figure 24. Bottlenose dolphin average abundance 2010–2015 (n= 2306, CV = 0.35). The map is based on Kernel density estimates showing zones with greater occurrence of bottlenose dolphins.

Based on the survey results, new Natura 2000 sites for harbour porpoise and bottlenose dolphin (pSIC Maceda – Praia da Vieira in the central part of western Portugal, pSIC Costa de Setúbal southeast of Lisbon) and new borders for existing sites were proposed.

In the Azores, photo-identification data of sperm whales collected opportunistically around the islands of Faial and Pico during the summer months of 2009–2015 were analysed with open Jolly-Seber models with individual covariates and multistate open robust design models to estimate the total number of female and subadult sperm whales summering in the study area in each year (annual abundance) and over the seven year period (abundance of the superpopulation). The annual population size of female and subadult sperm whales ranged from a minimum of 275 (95% CI: 188–404) individuals in 2014, to a maximum of 367 (95% CI: 248–543) individuals in

2012. The total population size of female and subadult sperm whales that summered in the study area between 2009 to 2015 was 1468 (95% CI: 1203–1791) individuals (Boys *et al.*, 2019).

UK: BOTTLENOSE DOLPHIN POPULATION ESTIMATES WITHIN CARDIGAN BAY, WEST WALES.

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 (SAC) 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 2018) were 121 (95% CI 99–185) for the Cardigan Bay SAC and 186 (95% CI 152–263) for the wider Cardigan Bay using closed population models. The equivalent estimates using robust open population models were 119 (95% CI 93–187) for the Cardigan Bay SAC and 150 (95% CI 137–184) for the wider Cardigan Bay. Over the 18-year period, population size has fluctuated both within Cardigan Bay SAC and the wider Cardigan Bay. The Capture–Mark–Recapture information indicates movements of animals between Cardigan Bay and areas to the north (North Wales, Liverpool Bay, and the Isle of Man). A comparison of movements of individuals of known gender indicated that the majority of mothers (91%) gave birth within Cardigan Bay, but only 69% remained in the area after giving birth (Duckett, 2018). This also supports the finding that there are nearly twice as many calves encountered in Cardigan Bay than in North Wales.

UK: HARBOUR PORPOISE USAGE OF TIDAL STREAM ENVIRONMENTS IN NORTH ANGLESEY, WALES

New renewable technologies are being considered to help the UK reach its target of delivering 15% of its energy consumption from renewable sources by 2020. Included amongst these is tidal energy. One such region being investigated is Anglesey in North Wales where strong currents flow around the north and northwest coasts. Harbour porpoises are common within these tidal stream environments; hence, studies to better understand their spatial and temporal occupancy of these habitats at a regional scale are conducted. One study has quantified and compared relationships between the presence of harbour porpoise and several hydrodynamic characteristics across four tidal stream environments (Waggitt *et al.*, 2017). Within those sites, encounters with animals were concentrated in small areas (<200 m^2) and increased during certain tidal states (ebb vs. flood). In sites showing relatively high maximum current speeds (2.67–2.87 ms-1), encounters were strongly associated with the emergence of shear-lines. In sites with relatively low maximum current speeds (1.70–2.08 ms-1), encounters were more associated with areas of shallow water during peak current speeds. The overall probability of encounters was higher in low current sites. It is suggested that the likelihood of interactions could be reduced by restricting developments to sites with high maximum current speeds (>2.5 ms1), and placing turbines in areas of laminar currents therein.

A second study in June–July 2018 at two sites in North Anglesey used direct visual observations, supplemented by a tripod-mounted camcorder, filming the movement of the individual animals (Blanchard, 2018). The recordings taken were then analysed and response variables related to surface features, and hydrodynamic models of turbulence, current speed, depth and tidal range developed for the two areas. Harbour porpoises at both sites showed movements suggestive of foraging during all tidal states, but primarily during the ebb and high slack tides at one site but during flood and low slack tides at the second site. Turbulence was shown to be a significant factor influencing behaviour and movement at the site with the greater occupancy, with porpoises showing concentrated movements of fast foraging in areas of high turbulence.

NORTH SEA:

Strandings of white-beaked dolphins on the North Sea coast were investigated in 1991–2017 (IJsseldijk *et al.*, 2018b). Stranding numbers in the northern North Sea have been stable throughout the study period, with a marginal increase in annual numbers over the study period. In the central North Sea, declining numbers from the early 1990s until around 2000 were followed by a slight increase. In the southern North Sea, a declining trend is evident during most of the time-series. The kernel estimated density distribution of stranded animals throughout the study period confirmed these trends. It showed a shift from higher densities in the southern part of the North Sea in 1991–1999 to increased densities in the northwestern area and decreased densities in the south in more recent times (Figure 25). This could be a first indication of a change in white-beaked dolphin distribution from southerly to more northerly regions.

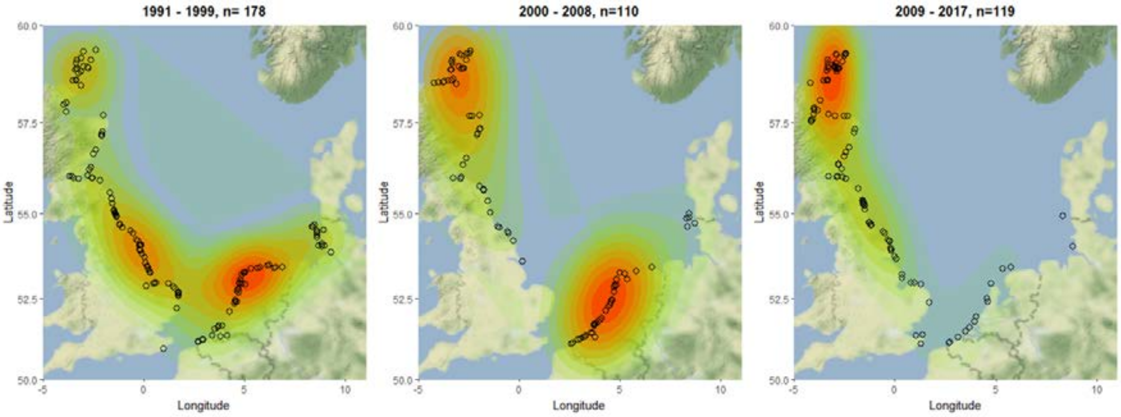


Figure 25. Density of white-beaked dolphin strandings in three periods (1991–1999; 2000–2008; 2009–2017) for the United Kingdom, Belgium, the Netherlands and Schleswig-Holstein, Germany (n = 407).

A stranding of 30 sperm whales in six countries around the North Sea in 2016 resulted in a flurry of papers, published in 2018. Smeenk and Evans (2018) provided a historical context for the 2016 strandings by collating sperm whale strandings around the North Sea from the 13th century to present day. Strandings are of all times, but an increase was evident since the 1980s. Several different processes may have contributed to the strandings, but Pierce *et al.* (2018) found correlations between strandings and temperature. The latter could affect sperm whales via their prey, predominantly the squid *Gonadus fabricii*.

IJsseldijk *et al.* (2018c) summarized the results of the studies performed on the majority of the stranded animals of 2016 (n = 27). All sperm whales were immature males. The animals were in fair to good nutritional condition and, aside from the pathologies caused by stranding, did not exhibit significant evidence of disease or trauma. Infectious agents were found, including various parasite species and several bacterial and fungal pathogens. Marine litter was found in nine of the sperm whales. However, none of these findings were considered to have been the primary cause of the stranding event.

The genetic diversity of the stranded sperm whales suggests that they originated from disparate regions/subpopulations throughout the entire Atlantic, rather than from a single area (Autenrieth *et al.*, 2018). While genetic data unravel relatedness and kinship, contamination data integrate over areas, where animals occurred during their lifetime. Combining the genetic data with contaminant data provided evidence of at least two cohorts with different origin among these stranded sperm whales; one from the Canary Island region and one from the northern part of the Atlantic (Schnitzler *et al.*, 2018).

NORTHWEST EUROPEAN SEAS: Continuing the 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 and Bangor University have collated around three million kilometres of cetacean survey effort from more than fifty research groups in Northwest European seas covering the period 1978–2018. 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 prey capture availability for different cetacean species, integrating the probability of encountering the species and its abundance, density maps of the 12 most common species have been produced at monthly temporal and 10 km spatial resolution across the past three decades. January and July summaries are shown in Figures 26 and 27. Seasonal patterns for each of the 12 most common species are shown in Figure 28.

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 (2017–2019), risk mapping is being undertaken where monthly distributions are compared with those of different human pressures (risk of bycatch from fishing activities using different gear types, and targeting different prey species; risk of collision from vessel strikes; and disturbance from underwater continuous and impulsive noise), incorporating measures of the different vulnerabilities that a species faces from a particular activity.

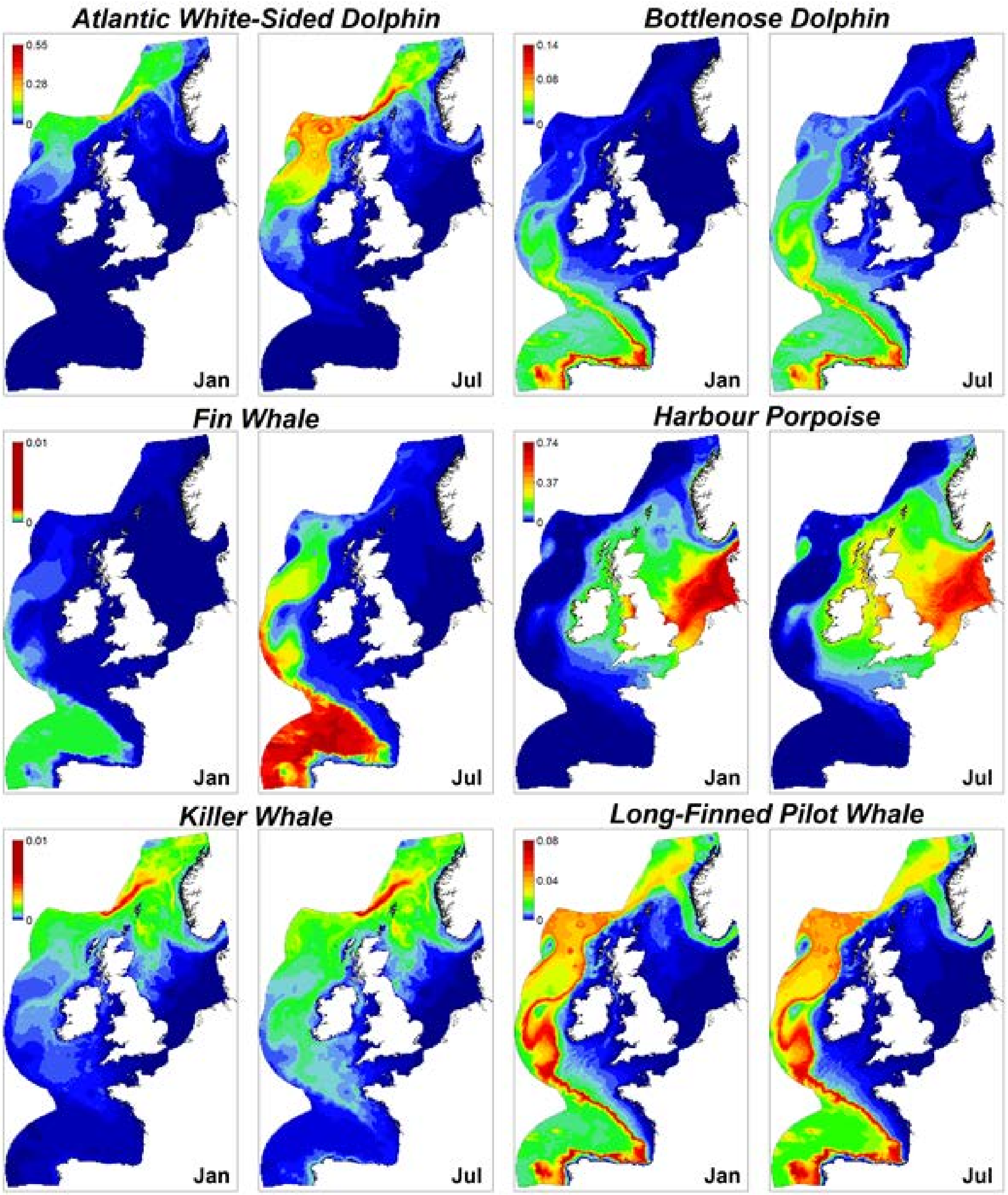


Figure 26. Predicted average January and July densities (animals per km²) for white-sided dolphin, bottlenose dolphin (offshore ecotype), fin whale, harbour porpoise, killer whale and long-finned pilot whale in the Northeastern North Atlantic. A different colour gradient is used for each species.

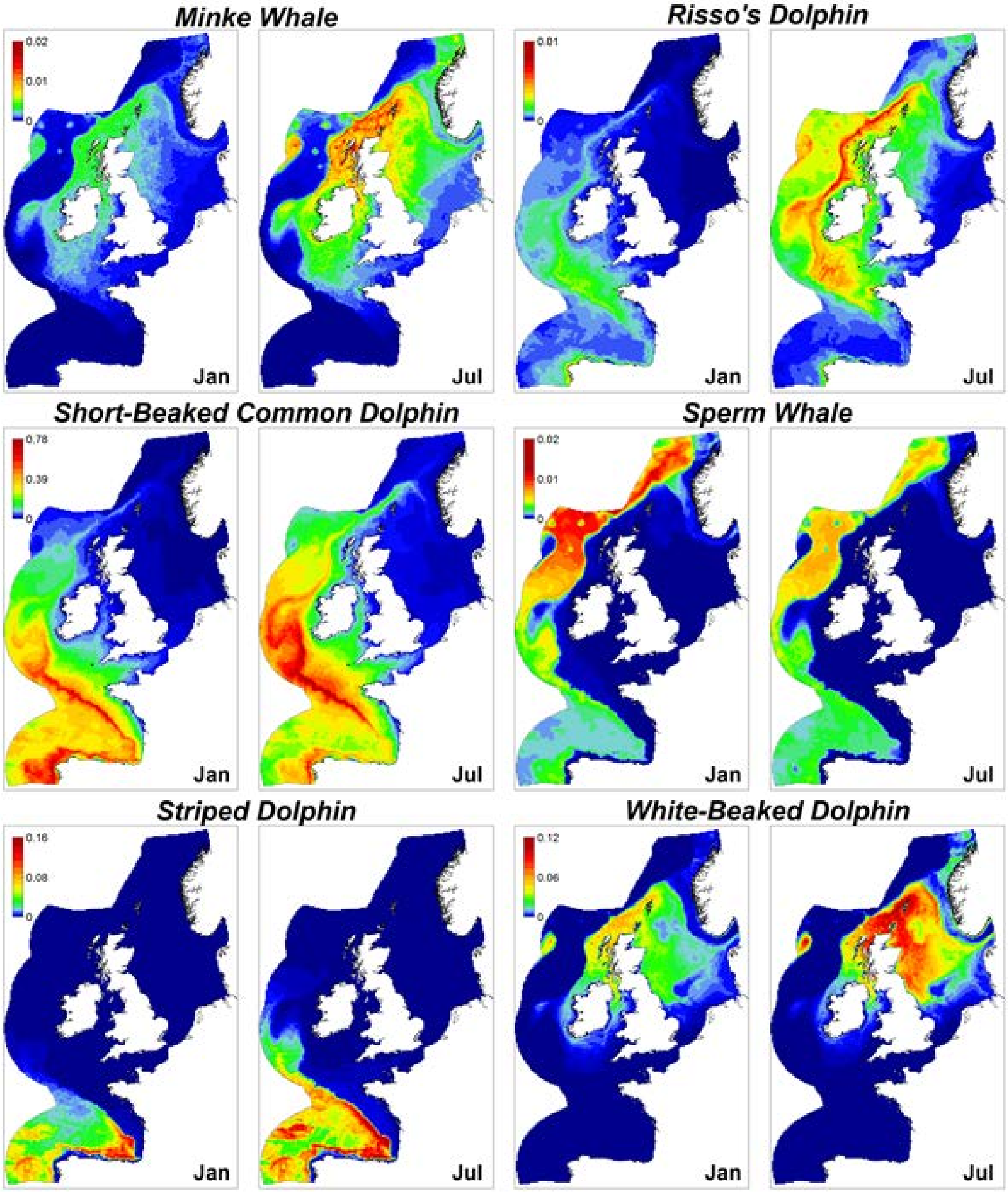


Figure 27. Predicted average January and July densities (animals per km^2) for minke whale, Risso's dolphin, short-beaked common dolphin, sperm whale, striped dolphin and white-beaked dolphin in the Northeastern North Atlantic. A different colour gradient is used for each species.

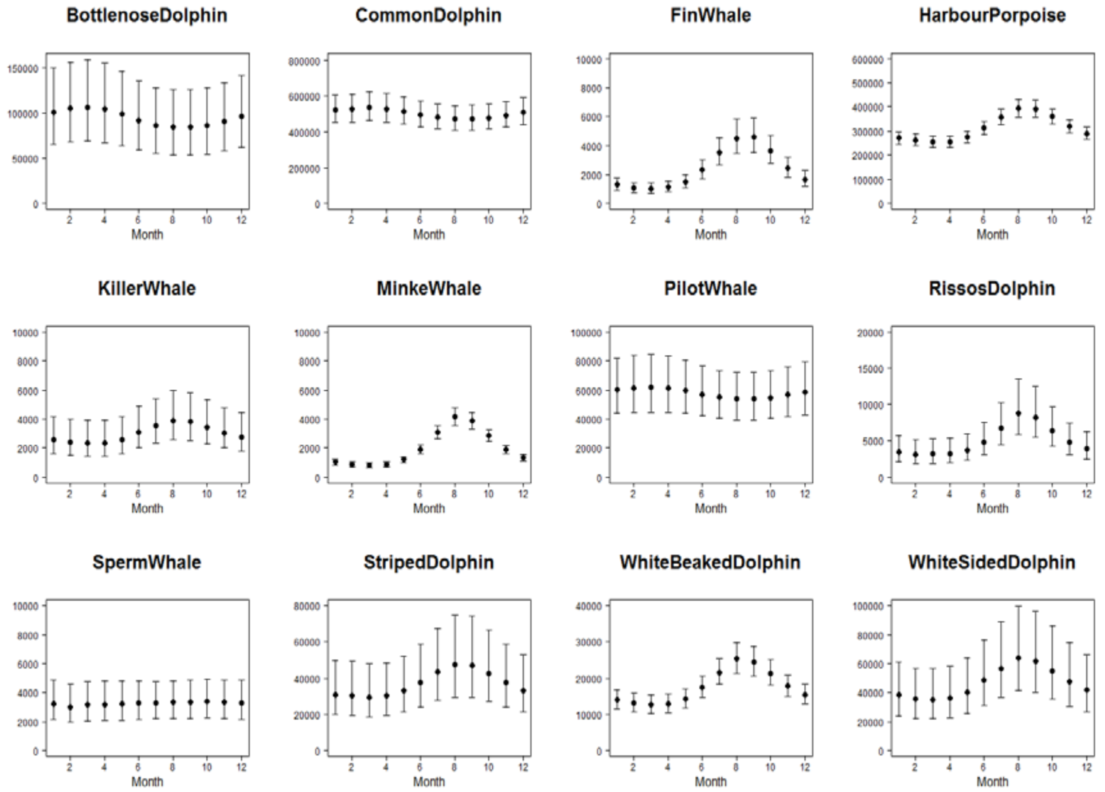


Figure 28. Predicted seasonal variation in abundance for 12 cetacean species in the Northeastern North Atlantic.

NORTH ATLANTIC OCEAN: Virgili *et al.* (2019) assembled visual ship-based and aerial surveys in the North Atlantic Ocean and the Mediterranean Sea between 1998 and 2015, totalling 1 240 000 km of effort. Relative densities of beaked whales, sperm whales and kogiids were calculated with generalized additive models. The reliability of the predictions outside the surveyed area was assessed by performing a gap analysis.

Beaked whales had high predicted relative densities of along steep slope areas associated with deep depths and high gradients of temperature, particularly on the western side of the Atlantic Ocean. The lowest relative densities were predicted in the Mediterranean Sea.

Predicted relative densities of sperm whales increased in deep waters (>2000 m) associated with high gradients of SST and high NPP. The highest relative densities were also predicted on the western side of the Atlantic Ocean, along the Gulf Stream, whereas they were lowest in the Mediterranean Sea.

The highest relative densities of kogiids were found in deep waters associated with fronts, canyons and seamounts. The highest relative densities were predicted on the western side of the Atlantic Ocean, along the Gulf Stream.

1.2 New information on population/stock structure

Bottlenose dolphin (*Tursiops truncatus*)

Using a combination of photo-identification and population genetics, Nykänen *et al.* (2018) quantified gene flow and demographic dispersal between SACs of bottlenose dolphins on the west coast of Ireland. Three populations were identified in the waters of western Ireland, two of which have largely non-overlapping core coastal home ranges and are each strongly spatially associated with specific SACs. High site fidelity of individuals was found within each of these two coastal populations to their respective SACs. Low levels of demographic dispersal between the populations was also found, but it remains unclear whether any mating occurs between populations through these migrants (genetic dispersal). The population sampled in the Shannon Estuary has a low estimated effective population size and appears to be genetically isolated.

On a larger spatial scale, using a common set of microsatellite markers, Nykänen *et al.* (in revision) found a high level of genetic differentiation between coastal and pelagic populations along the European Atlantic margin. Genetic structure was defined at an unprecedented fine-scale level for coastal dolphins leading to identification of five distinct coastal populations inhabiting the following areas: Shannon estuary, west coast of Ireland, English Channel, coastal Galicia, east coast of Scotland, and Wales/West Scotland. Demographic connectivity was very low among most populations with less than 10% migration rate suggesting no demographic coupling among them. Each local population should therefore be monitored separately.

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

WGMME has in previous years reported 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; 2017; 2018). No new information to report.

1.4 New information on anthropogenic threats

A framework to assess cumulative effects on cetaceans is discussed in ToR D.

1.4.1 Fishery bycatch

New information not covered in ToR C includes:

NETHERLANDS: Scheidat *et al.* (2018) assessed the bycatch of harbour porpoise in the Dutch commercial bottom-set gillnet fishery using Remote Electronic Monitoring (REM). From 1 June 2013 to 31 March 2017, fourteen fishing vessels were equipped with closed-circuit television cameras (CCTV) in combination with sensors to obtain video footage, time and position of all net hauls. Video footage was analysed for 900 fishing days (of a total of 8133 fishing days of the overall fleet). Bycatch rate (porpoises/net length km) was different between net types, with 0.004 for trammelnets and 0.0006 for single-walled gillnets. The bycatch rate for all net types combined (0.0011) was applied to calculate bycatch numbers, resulting in an estimate of 88 animals for the complete study period (95% C.I. 6–170; CV = 14.54) and an annual average of 23 animals (95% C.I. 2-44). The scale of the average annual mortality for the 'Dutch' porpoise population was assessed to be between 0.05 and 0.07% (for the study period). Key recommendations from this study were the assessment of all causes of bycatch in Dutch waters, the development of a cost-effective mobile REM system, continuation of REM within the Data Collection Framework, facilitation of CITES permits to land animals and the improvement of the data collection methodology of fishing effort on an international level.

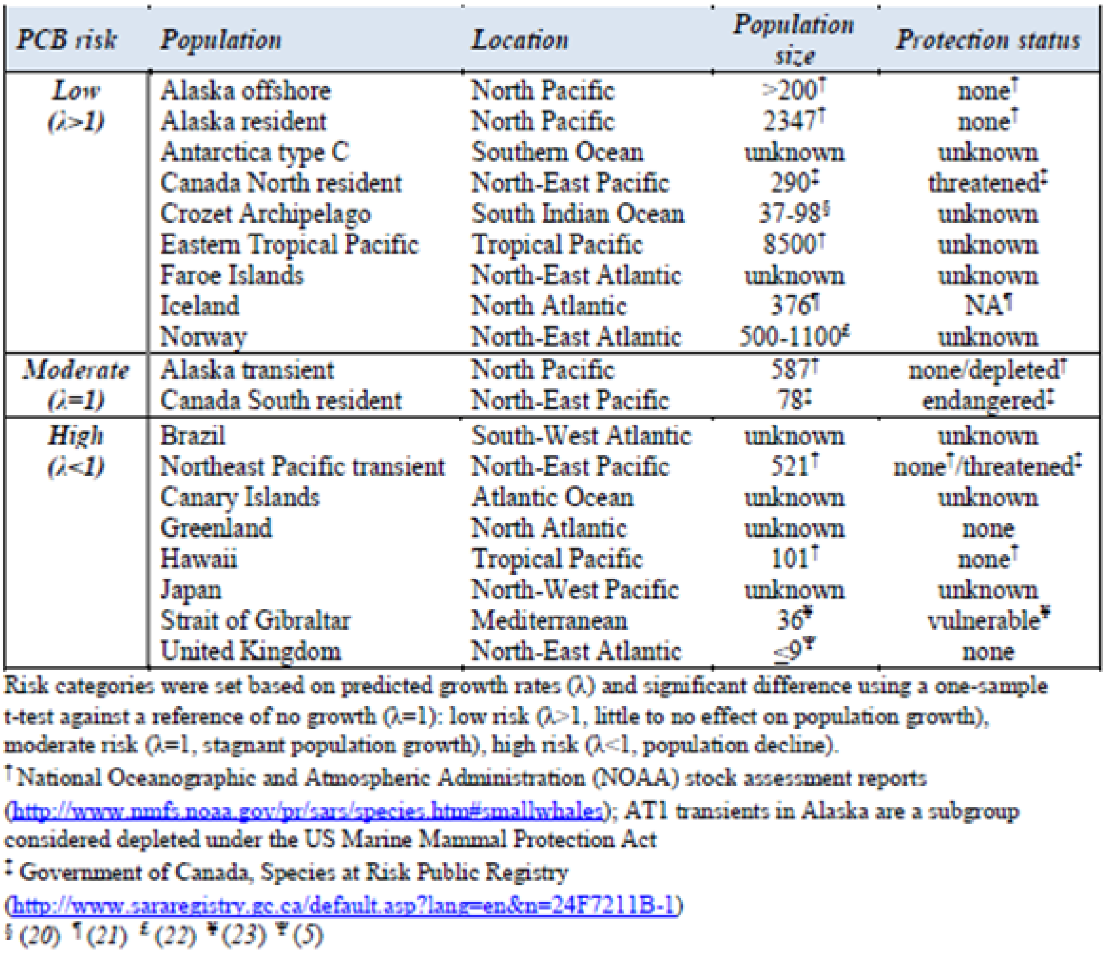
PORTUGAL: The LIFE+ MarPro project (Vingada and Eira, 2018) provided estimates of accidental mortality in fisheries obtained from surveys, observers on board fishing vessels, electronic monitoring systems or through voluntary declarations from fishermen and strandings analyses, for the first time in Portugal. This resulted in the following estimates, for the period 2010–2015 (Vingada *et al.*, 2015):

- The purse-seine fleet captured an average of 321 cetaceans per year, contributing to the removal of 1.08% of the harbour porpoise population, 0.64% of the common dolphin population and 0.72% of bottlenose dolphin population. The PBR values are below the limit of 1.7%.
- The polyvalent fleet captured a total of 3639 cetaceans per year, assuming a minimum effort scenario (10% of the national fleet does not operate annually, average 180 days at sea). It contributes to the removal of 13.24%, 7.34% and 5.10% of the harbour porpoise, common dolphin and bottlenose dolphin populations, respectively. For all species, PBR values are well above the limit of 1.7%.
- The bottom-trawler fleet captured an average of 437 cetaceans per year. It contributes to the removal of 0.92% of the common dolphin population and 1% of the Bottlenose dolphin population. For all species with catches, PBR values are below the 1.7% limit.
- The deep-sea longline fleet captured an average of 231 cetaceans per year. This fleet contributes to the removal of 0.10% of the common dolphin population, 0.44% of striped dolphin population and 0.70% of the bottlenose dolphin population. For all species with captures, PBR values are below the limit of 1.7%.
- The beach-seine fleet (Xávega) captured on average 47 cetaceans per year. Xávega contributes to the removal of 1.4% of the harbour porpoise population and 0.06% of the common dolphin population. PBR values are below the 1.7% limit. However, considering that beach-seine is a seasonal activity operating only in some areas of the country, values for harbour porpoise are considered very high.

1.4.2 Pollution: persistent organic pollutants and toxic elements

A recent paper by Desforges *et al.* (2018) assessed PCB effects on global killer whale populations, which are among the most highly contaminated mammals in the world. Using an Individual Based Model framework and globally available data on tissue PCB concentrations in killer whales, showed that PCB effects on reproduction and immune function threaten the long-term viability of more than 50% of the world's killer whale populations. PCB mediated effects over the coming 100 years predicted that killer whale populations near ten industrialized regions, and those feeding at high trophic levels regardless of location, are at high risk of population collapse. Those regions include the Canary Islands, Straits of Gibraltar, and the UK (Table 13). Despite their near global ban over 30 years ago, PCBs continue to devastate marine ecosystems (Desforges *et al.*, 2018).

Table 13. Global assessment of population-level risk from PCB exposure. Taken from Desforges *et al.* (2018).



PCB risk	*Population*	*Location*	*Population size*	*Protection status*
Low *(λ>1)*	Alaska offshore	North Pacific	>200†	none†
	Alaska resident	North Pacific	2347†	none†
	Antarctica type C	Southern Ocean	unknown	unknown
	Canada North resident	North-East Pacific	290‡	threatened‡
	Crozet Archipelago	South Indian Ocean	37-98§	unknown
	Eastern Tropical Pacific	Tropical Pacific	8500†	unknown
	Faroe Islands	North-East Atlantic	unknown	unknown
	Iceland	North Atlantic	376¶	NA¶
	Norway	North-East Atlantic	500-1100£	unknown
Moderate *(λ=1)*	Alaska transient	North Pacific	587†	none/depleted†
	Canada South resident	North-East Pacific	78‡	endangered‡
High *(λ<1)*	Brazil	South-West Atlantic	unknown	unknown
	Northeast Pacific transient	North-East Pacific	521†	none†/threatened‡
	Canary Islands	Atlantic Ocean	unknown	unknown
	Greenland	North Atlantic	unknown	none
	Hawaii	Tropical Pacific	101†	none†
	Japan	North-West Pacific	unknown	unknown
	Strait of Gibraltar	Mediterranean	36¥	vulnerable¥
	United Kingdom	North-East Atlantic	≤9Ŧ	none

Risk categories were set based on predicted growth rates (λ) and significant difference using a one-sample t-test against a reference of no growth (λ=1): low risk (λ>1, little to no effect on population growth), moderate risk (λ=1, stagnant population growth), high risk (λ<1, population decline).

† National Oceanographic and Atmospheric Administration (NOAA) stock assessment reports (http://www.nmfs.noaa.gov/pr/sars/species.htm#smallwhales); AT1 transients in Alaska are a subgroup considered depleted under the US Marine Mammal Protection Act

‡ Government of Canada, Species at Risk Public Registry (http://www.sararegistry.gc.ca/default.asp?lang=en&n=24F7211B-1)

§ *(20)* ¶ *(21)* £ *(22)* ¥ *(23)* Ŧ *(5)*

UK: A recent study assessed for evidence of reproductive failure and reproductive dysfunction in female common dolphins inhabiting UK waters that may be possibly linked to exposure to PCBs (Murphy *et al.*, 2018), following criteria laid out in Murphy *et al.* (2015). Results of the study which assessed samples collected between 1990 and 2013, suggested that reproductive failure could have occurred in 30% or more of mature females in a control sample (stranded dolphins that were identified as bycatch mortalities from necropsies; n = 23). Reported incidences of reproductive dysfunction are rare in cetaceans, however within a large sample of bycaught and other stranded females (control and non-control samples), 16.8% (18 out of 107) presented with reproductive system pathologies including conditions such as vaginal calculi (5.6%), suspected precocious mammary gland development (5.6%), and ovarian tumours (2.8%). Individual females also presented with an ovarian cyst, atrophic ovaries in a sexually immature individual and the first reported case of an ovotestis in a cetacean species (Murphy *et al.*, 2011; 2018). Where pollutant data were available, all observed cases of reproductive tract pathologies in both control and non-control samples were reported in females with ΣPCB burdens >22 mg/kg lw. However, combined effects from exposure to multiple pollutants, including (low doses of) DDT and other legacy and emerging pollutants, were not ruled out, and requires further investigation (Murphy *et al.*, 2018).

Murphy *et al.* (2018) applied toxicity thresholds to all available ΣPCB data for common dolphins in the Northeast Atlantic that were sampled between 1990 and 2013 (n = 183). 76% of sexually immature males and females had ΣPCB levels above the 9 mg/kg threshold for onset of adverse health effects in marine mammals, and 17% had levels greater than one of the highest toxicity thresholds for marine mammals, 41 mg/kg. ΣPCB ranged from 1.1 to 95.9 mg/kg in sexually immature individuals. Fifty percent of mature males had blubber ΣPCB concentrations above the 41 mg/kg threshold for profound reproductive effects in female seals. Mean ΣPCB was 45.8 mg/kg in mature males (who are unable to offload their pollutant burden), and concentrations ranged from 7.0 to 119.8 mg/kg lw: the highest ΣPCB concentrations were observed in a male stranded in 1992.

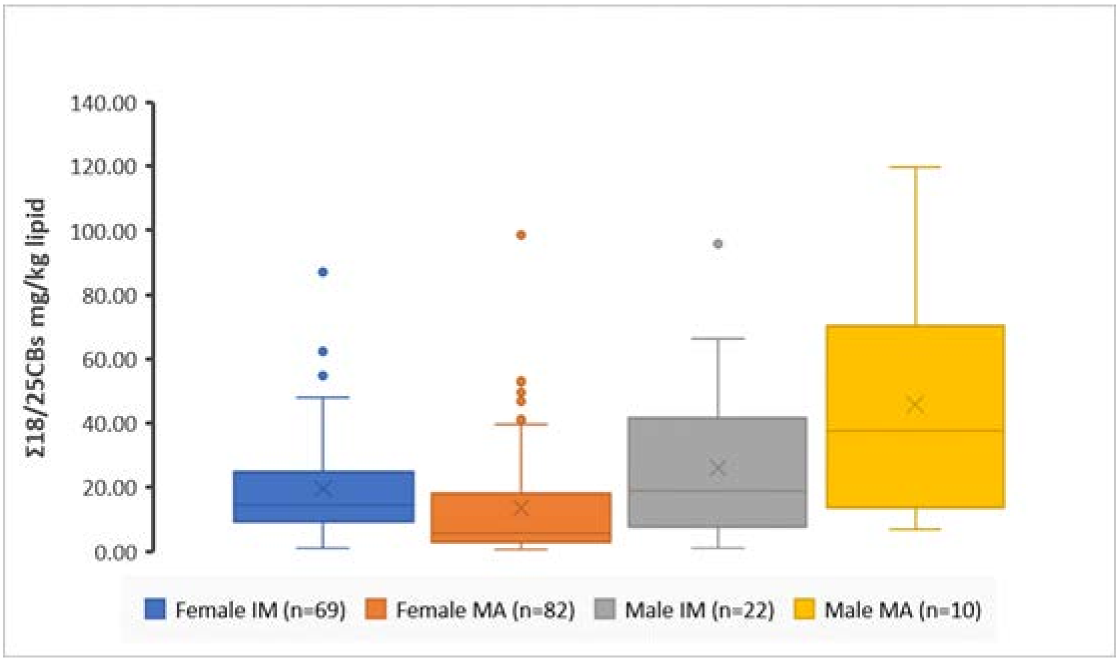


Figure 29. Box plots of male and female common dolphin reproductive status (IM = sexually immature, MA = Sexually Mature) and ΣPCB from stranded and bycaught common dolphins (1990–2013, n = 183). The dark horizontal line indicates the median, x-markers indicate the mean and outliers are highlighted by circles. Figure taken from Murphy *et al.* (2018).

Within the UK, the largescale NERC funded "ChemPop" project is investigating statistical trends in a range of Persistent Organic Pollutants (POPs) and their exposure and associated risks in UK cetaceans (harbour porpoises, bottlenose dolphins and killer whales). The project partners include the Centre for Ecology and Hydrology, the Institute of Zoology London, University of Hull and Brunel University London, and the project will run between 2018 and 2022. The UK Department of the Environment, Food and Rural Affairs is funding a second project that is undertaking analysis of PCBs in UK-stranded species that have not been assessed in detail to date; including the common dolphin, striped dolphin, Atlantic white-sided dolphin, white beaked dolphin, Risso's dolphin and long-finned pilot whale. This project is in collaboration with UK CSIP/SMASS and will also fund extension to the harbour porpoise time-series (2016 and 2017), and recent samples obtained from stranded killer whales and bottlenose dolphins (2016–2017). Analyses will also assess blubber and muscle samples in harbour porpoises, to investigate the potential impact of lipid mobilisation.

IRELAND: A recent study has commenced in Ireland, undertaken at the Galway-Mayo Institute of Technology, that is analysing legacy pollutants such as polychlorinated biphenyls, organochlorine pesticides and brominated flame retardants and heavy metals as well as emerging pollutants such as neonicotinoids in a range of cetacean species, including both odontocetes and mysticetes. Previously the Marine Institute analysed mammal blubber tissue samples on an ad hoc basis for organochlorines and a range of other pollutants, which for many analyses the results have not been published to date.

FRANCE: Work was recently undertaken assessing persistent organic pollutants (POPs) and mercury (T-Hg) in the blubber and skin, respectively, of free-ranging bottlenose dolphins from the Normanno-Breton Gulf that were sampled between 2010 and 2012 (Zanuttini *et al.*, in press). Among all the POPs analysed in the study, the ΣNDL-PCBs (non-dioxin like PCBs) were the most abundant compounds found in the blubber, followed by ΣDDX > ΣDL-PCBs > ΣPBDEs > dieldrin > Σendosulfan > HCB > ΣHCHs > Σchlordane > ΣPCDFs > ΣPCDDs. Mean concentrations of the ΣNDL-PCBs were 132 940 and 64 504 ng.g-1 lipid weight (lw) for males and females, respectively (Zanuttini *et al.*, in press). Among the NDL-PCBs, the hexachlorobiphenyls (PCB 153 and PCB 138) were the major compounds (ranging from 64 to 80%), followed by the heptachlorobiphenyls (PCB 180). Within the study 57 bottlenose dolphins (out of 58) exceeded the 9 mg/kg threshold (as ΣPCBs, see earlier text), and 51 bottlenose dolphins (out of 58) exceeded the higher 41 mg/kg threshold (as ΣPCBs, see earlier text).

Analysis of temporal trends in organic contaminants in harbour porpoises in French waters is currently being undertaken. Samples from 67 male porpoises have been processed, and individuals were sampled between 2001 and 2017. All males were assumed to be mature, though age determination is ongoing. 69.7% of porpoises showed PCB concentrations above Kannan's toxicity threshold of 17 µg/g lipid for total PCBs (as Aroclor 1254). An increase in PCB concentrations was observed during the time period, though this has not yet been tested statistically (Figure 30).

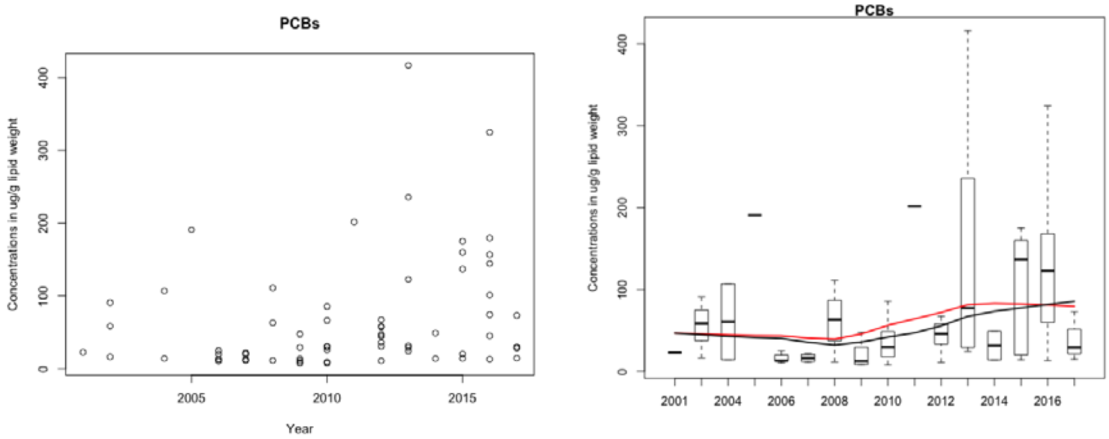


Figure 30. PCB concentrations in 67 male porpoises from France sampled between 2001 and 2017. (Paula Mendez Fernandez, unpublished data).

OTHER COUNTRIES: A marine mammal blubber PCB toxicity indicator was proposed to OSPAR by the ICES WGMME in 2013 for inclusion as a biodiversity common indicator within the MSFD (ICES, 2013). In 2019, the OSPAR Marine Mammal Expert Group under COBAM further developed this indicator for inclusion within EU MSFD Descriptors 1 (Biodiversity) and D8 (Contaminant effects). This proposed indicator was submitted to both, OSPAR Biodiversity Committee (BDC) and Hazardous Substances & Eutrophication Committee (HASEC), in March 2019.

A number of chemical pollutant studies are currently ongoing in several countries in western European waters. Analyses are ongoing, and this will be reported on in ICES WGMME 2020.

1.4.3 Marine debris

A workshop on marine litter was held during the annual conference of the European Cetacean Society in March 2017 (Panti *et al.*, 2019). Although this workshop was focused on marine litter, the outputs highlighted that information on diet, habitat, and pathological condition should be used to understand the sources, transfer and effects of marine litter, and therefore their impacts on marine mammals. The workshop concluded that further research and standardization of protocols are essential to understand these impacts. The workshop recommended a threefold approach to assess the impact of litter on marine mammals (Fossi *et al.*, 2018).

1. Analysis of gastro-intestinal content: Detection of the occurrence and rate of marine litter ingestion and any associated pathology through analysis of the gastro-intestinal content (with a particular focus on plastics and microplastics) in stranded cetaceans;
2. Analysis of the levels of plastic additives, as a proxy for ingestion: The plastic additives indirect quantification can be applied both to free-ranging as well as to stranded organisms. The levels of plastic additives (such as phthalates or polybrominated diphenyl ethers (PBDE)) and associated Persistent Bioaccumulative and Toxic (PBT) compounds allow evaluation of the exposure to marine plastic pollution.
3. Analysis of biomarker responses: Biological responses can be used to detect the potential toxicological effect related to PBT and plastic additives related to plastic ingestion in free-ranging individuals or in stranded organisms up to a few hours after death.

GERMANY: For a pilot project (2018–2020) on microplastics in marine mammals, hindgut including faeces (since 2014) and stomach (from 2019 onwards) samples from harbour porpoises, harbour seals and grey seals were taken according to an established protocol for avoiding secondary pollution and conservable loss, and stored in glass jars at -20°C. In addition, faeces samples from seals collected on sandbank haul-outs will be incorporated in this study. For sample handling, the protocol by Bravo Rebolledo *et al.* (2013) is used and adjusted to serve the demand of isolating microplastic particles from the different samples of all three species. Furthermore, in order to minimise losses and reduce secondary pollution, an established working area is used within a specifically created acrylic box. After isolation, the microplastic particles will be assessed and categorised under a binocular. The final identification of the polymer composition of each particle will be implemented via RAMAN spectroscopy.

UK: Marine debris has been assessed in the whole digestive tracts of 50 individuals from ten marine mammal species stranded along the UK coasts (Nelms *et al.*, 2019). The examination was carried out whilst operating strict contamination controls. The species were Atlantic white sided dolphin (1), bottlenose dolphin (1), common dolphin (16), grey seal (3), harbour porpoise (21), harbour seal (4), pygmy sperm whale (1), Risso's dolphin (1), striped dolphin (1), white-beaked dolphin (1). In total, 273 particles were detected, whereof 261 were less than 5 mm in size, i.e. classified as microplastics. Microplastic particles were detected in every animal examined (mean ± SD: 5.5 ± 2.7 particles per animal; range 1–12). Of the microplastics, 84% were fibres and 16% were fragments. Only one animal was found to contain macroplastics; green netting in the forestomach of a juvenile common dolphin. The abundance of microplastics was higher in the stomach(s) (3.8 ± 2.5) than in the intestines (1.7 ± 1.4), indicating that the stomach(s) acts as an entrapment site within the digestive tract, partially retaining the microplastics. It was examined whether taxon, age class, sex, length, cause of death was associated with the microplastic burden. Model simplification indicated that cause of death was the only statistically significant predictor of microplastic abundance ($p = 0.01$). Animals that had died of infectious diseases contained slightly higher mean microplastics abundances (7.0 ± 2.7), followed by trauma (4.7 ± 2.1) and other (4.6 ± 3.2). The same pattern was observed within harbour porpoises and common dolphins, the only species with a sample size of more than 16 individuals. A possible biological significance of this pattern is not yet known. As all investigated species are raptorial feeders, it was considered most likely that the microplastics had been consumed indirectly through trophic transfer from contaminated prey.

SPAIN: Microplastics were investigated in the stomachs of 35 common dolphins stranded along the Galician coast of the Iberian Peninsula during 2005–2010 (Hernandez-Gonzalez *et al.*, 2018).

A total of 411 particles were found, whereof one was a fibre just above 5 mm in length and the rest smaller. In total 96.6% of the particles were fibres (including the fibre just above 5 mm), 3.2% fragments, and 0.2% a bead. Microplastics were found in every animal examined (mean ± SD: 12 ± 8 particles per animal; range 3–41). Applying a cut-off value of 2.3 mm to exclude possible airborne particles, a common dolphin would have a 94% probability of having microfibres in its stomach contents with an average number of 3.6 fibres with an average size of 3.73 mm. Total length or sex were not found to correlate with the number of microplastics found per individual, however there was a negative trend over the years 2005–2010. Several possible factors may explain the difference over the years, such as changes in feeding areas or diet.

1.4.4 Underwater noise

DENMARK: Clausen *et al.* (2018) investigated the impact of ambient noise on the performance of two different Passive Acoustic monitoring (PAM) devices and different harbour porpoise click train detection algorithms, in terms of their ability to accurately classify porpoise click trains under changing ambient signal-to-noise ratios. Two different detector types (C-POD click detector [Chelonia Ltd., Penzance, UK] and SM2M+ full bandwidth recorder [Wildlife Acoustics, Boston, USA]) were deployed near an offshore oil and gas production platform in the Danish North Sea for 42 days, collecting both harbour porpoise echolocation clicks and ambient noise data in an environment where ambient noise levels varied considerably over short time periods due to waves, weather, platform activity, shipping, etc. Data from both device types indicated that porpoise click detection rates declined with increasing noise levels, which needs to be considered when using such data to underpin conclusions about potential noise impacts on porpoises and other cetaceans. The authors point out that increasing ambient noise levels may also result in changes to porpoises' behaviour and sound outputs (e.g. displacement, reduced vocalisation rates, increasing source levels), all of which could result in either reduced or inflated detection probabilities that may not be fully accounted for. Clausen *et al.* (2018) conclude that PAM cues „...should be analysed with care if they are to be used to determine the effect of noise on porpoise presence, as several other factors, most notably the detection process itself, can be affected by the noise.", which is likely to be relevant to other acoustic impact studies involving cetaceans.

DENMARK/GREENLAND: Kyhn *et al.* (2019) quantified the contributions of four concurrent seismic surveys, which occurred in northeastern Baffin Bay, off the Greenland coast, in August-October 2012, to the local soundscape, with a view to evaluating potential cumulative impacts on local marine mammals. Data were collected using a combination of close-range (Reson TC4032/TC4034 hydrophones, attached to an Olympus LS-11 recorder, deployed from a vessel out to 15 km from source) and long-range sound recorders (automated DSG-Ocean units, Loggerhead instruments, Sarasota, Florida, USA, deployed at seven moorings across Baffin Bay, as well as similar, independent data collection efforts by JASCO on behalf of Shell (Martin *et al.*, 2017)). Data were analysed to characterise airgun pulses, quantify long-range propagation of airgun signals and their contribution to ambient noise levels, comparing empirical observations to existing sound transmission model outcomes. Results indicated considerable increases to ambient noise levels in Baffin Bay as a direct result of seismic survey operations. The impact was cumulative as the noise level rose in response to the onset of each survey: on a minute-by-minute scale the sound-exposure-levels varied by up to 70 dB (20 dB on average), depending on range to the seismic vessel, local bathymetry effects and interference patterns, representing a significant change in the auditory scene for marine mammals. Airgun pulse energy did not decrease to ambient before arrival of the next pulse, leaving very little low frequency masking free time. Overall, the measured values matched well with preseason-modelling, emphasizing the importance of noise modelling in impact assessments, assuming that responses of focal marine mammals are understood. The authors highlighted particular risks of potential masking effects (particularly affecting bowhead whales), hearing damage (particularly affecting baleen whales at close ranges to source) and behavioural effects (affecting different cetacean and pinniped species in varying ways that remain poorly understood) of such increases in ambient noise levels on marine mammals.

GERMANY: Mikkelsen *et al.* (2018) reported on the deployment of long-duration acoustic and 3D-accelerometer tags (DTAGs) on three harbour seals and two grey seals in German waters. The aim of this study was to gather information on anthropogenic noise levels, mainly related to shipping, that are experienced by these animals, and to understand their potential responses to the noise. DTAGs were deployed for periods of up to 21 days, during which sound, accelerometry, magnetometry and water pressure were recorded continuously. Tagged seals spent between 6.6%–42.3% of their time hauled-out, and 5.3%–12.4% of their time resting near the seabed; the rest of the time was spent travelling, foraging and resting near the surface. Animals were exposed to audible vessel noise 2.2%–20.5% of their time when in water; numerous cases of apparent interruption of functional behaviours (e.g. resting), coinciding with high-level vessel noise were observed. In the case of one particularly detailed record from one of the harbour seals, only about two-thirds of the detected ship noise events could be linked to AIS vessel tracking system data, highlighting the potentially significant contribution to ambient shipping-related noise emissions by smaller vessels without AIS. The authors suggest that the combination of animal-borne sound and movement loggers demonstrated in this study may become increasingly significant as a future tool to assess disturbance effects and long-term population-level impacts of noise associated with marine anthropogenic activities.

NETHERLANDS: After 15 years of TTS studies to get insight into the effects of loud sounds in the oceans (pile driving, seismic surveys, military sonars and explosions) on hearing of marine mammals, the hearing studies from Kastelein slowly start to focus on masking due to chronic underwater sound such as shipping sound. Also, attention is focusing on behavioural responses of marine mammals both in the field and in captivity, in order to construct acoustic dose-behavioural response relationships. In addition, the effect of sound on populations (the ultimate goal) is getting more attention. These investigations require the collaboration of several disciplines: acousticians, modellers, physiologists studying energetics, and ecologists studying population dynamics. A recent study by Kastelein *et al.* (2018) investigated changes to swimming speeds of a harbour porpoise exposed to playbacks of underwater pile driving sounds (SPLs ranging from 130 to 154 dB re 1 μPa), based on video recordings originally described by Kastelein *et al.* (2013). All exposure experiments were conducted in a specially constructed outdoor pool. Compared to baseline mean swimming speeds of 4.3 km/h, the porpoise swam progressively faster when exposed to higher SPL, up to 7.1 km/h at SPL of 154 dB re 1 μPa. There was no evidence of increased respiration rates during the experimental period, suggesting that the animal could maintain this speed of approximately 7 km/h with relative ease for at least the duration of the experiments (30 minutes). This swim speed is higher than the speed of 5 km/h found by (Otani *et al.*, 2001). Considering the duration of individual pile driving events (approximately two hours), Kastelein *et al.* (2018) suggest that a porpoise located adjacent to such an event could theoretically end up swimming ca 14 km in an attempt to flee the high sound levels. This avoidance distance is lower than avoidance distances between 17–21 km found in studies on Danish and German windfarms (Brandt *et al.*, 2011; 2016; Dähne *et al.*, 2013; Tougaard *et al.*, 2009), but it falls in the range of 10–20 km avoidance distance as shown in a Dutch study (Geelhoed *et al.*, 2018a). Avoiding pile-driving or other anthropogenic sound potentially leads to increased energy expenditure and heightened foraging requirements as a result of this disturbance.

Sertlek *et al.* (2019) undertook a comprehensive cumulative assessment of the soundscape in the Dutch sector of the North Sea, considering various anthropogenic (ships, seismic airguns, explosions [e.g. related to detonation of WW2-era ordnance]) and natural (wind) sound sources across a range of spatio-temporal scales and different frequency bands. Sound source data and propagation models were integrated to generate sound maps for the Dutch North Sea, both for individual sound sources and for all sources combined. Results indicated that most acoustic energy was derived from shipping, followed by airguns and explosions, particularly at frequencies<10 kHz. The contribution from anthropogenic sound sources to the overall sound energy budget in the Dutch North Sea was found to exceed that of wind by a factor of one hundred, reinforcing the evident importance of human activities in this region. The authors also highlighted the importance of considering shorter averaging times than is typical (i.e. seconds to hours, rather than days, weeks or years), when seeking to assess the significance of noise pollution to sensitive species such as cetaceans. This is especially important for transient and irregularly occurring sounds, such as explosions.

UK: Findlay *et al.* (2018) provided the first in-depth spatio-temporal analysis of the presence of Acoustic Deterrent Devices (ADDs) along the west coast of Scotland. ADDs are widely used in salmon aquaculture in Scottish waters to deter seals from fish farms (Quick *et al.*, 2004; Northridge *et al.*, 2010; 2013), but their environmental impact and long-term effectiveness remain unclear. The majority of these devices produce sounds in the range of 2 to 40 kHz, with source levels ≥185 dB re 1 μPa @ 1m (RMS; Gordon and Northridge, 2003; Lepper *et al.*, 2014; Reeves *et al.*, 2001). Although widely used in the Scottish salmon aquaculture sector, information on numbers or types of ADDs used at aquaculture sites is limited, and no license is required to use these devices (Coram *et al.*, 2014). Findlay *et al.* (2018) used acoustic point sample data („listening events") collected during cetacean line-transect surveys to map ADD presence between 2006 and 2016. A total of 19 601 listening events occurred along the Scottish west coast, and ADD presence was recorded during 1371 of these events. Results indicated a steady increase in ADD detections from 2006 (0.05% of listening events) to 2016 (6.8% of events), with the largest number of detections in 2013 (12.6% of events), as well as substantial geographic expansion. This study demonstrates that ADDs are a significant and chronic source of underwater noise on the Scottish west coast. Although not explicitly addressed by Findlay *et al.* (2018), such noise could have potential adverse impacts on both target (seals) and non-target (cetaceans) species, and improvements to monitoring and regulatory approaches are suggested.

Hastie *et al.* (2018) measured the behaviour of harbour seals in western Scotland (UK) responding to acoustic playbacks of simulated tidal turbine sounds within a narrow coastal channel subjected to strong tidally driven horizontal currents. In summer, this channel is frequented by ~100 harbour seals that haul-out on rocks along the channel shoreline and forage intensively within the channel. A total of ten seals were equipped with GPS tags which logged the location of the seals every time they surfaced. The channel was experimentally ensonified through an acoustic playback setup transmitting a simulated tidal turbine signal, which was based on recordings of the 1.2 MW tidal energy converter (SeaGen) which until recently operated in the narrow channel connecting Strangfor Lough (Northern Ireland) to the Irish Sea (Robinson and Lepper, 2013). The broadband RMS source level of the playbacks was 175 dB re 1 μPa-m(RMS), which was designed to reflect the estimated RMS source level of the real turbine (174 dB re 1 μPa-m(RMS)) (Robinson and Lepper, 2013). Visual observations from an adjacent clifftop location were used to collect data on relative abundance of seals in the channel. Changes in seal abundance and distribution within the channel were analysed using a generalized additive model framework. Results indicated that overall numbers of seals in the channel did not change significantly during playback periods; on the other hand, seals did exhibit localised spatial avoidance of the ensonified area, out to ranges of approx. 500 m from the playback setup. This effectively reduced the usage by seals within these ranges by 11–41% at the playback location, implying that the noise of the simulated turbine caused seals to modify their behaviour. This behavioural change would potentially have resulted in reduced risks of seals colliding with a real tidal turbine; however, further observations of animals in the vicinity of real turbines are now urgently required.

Merchant (2019) provided an in-depth review of potential noise reduction (abatement) measures that could be applied in various maritime industries, within current economic and policy contexts. The review illustrated the diversity of potential solutions to anthropogenic noise emissions by presenting different technological, operational and activity reduction measures for three marine industries that are collectively responsible for the majority of noise outputs (shipping, offshore wind farm construction, and seismic surveys for subsea hydrocarbon deposits, respectively). Measures included technological innovations (e.g. development of bubble curtains for pile driving, vibroseismic systems as an alternative to seismic airguns), changes to standard operations (e.g. implementation of ship speed restrictions) and development of new, stricter regulations (e.g. setting international standards for ship noise outputs, similar to existing measures governing the aviation sector). For each industry, the review identified various combinations of measures that could be applied by policymakers to effectively reduce noise outputs by these and other marine industries, thereby significantly reducing the acoustic impact of these sectors on marine mammals and other species.

Pine *et al.* (2019) assessed whether, and to what extent, the noise produced by marine tidal energy generators might result in auditory masking of natural auditory cues to marine mammals, specifically harbour seal and harbour porpoise. Listening Space Reductions (LSR; method described by Hannay *et al.*, 2016) were calculated for harbour seal and harbour porpoise, comparing summer and winter scenarios for two tidal energy devices of different designs. The effects of turbine noise on the listening space varied between the type of turbine, the species, the season and the ambient sound conditions within both seasons. Higher masking impacts, in terms of LSRs, were seen for harbour seals, with averaged LSRs exceeding 90% within 62 m from the turbine, compared to the maximum averaged LSR of 71% within 10m range for harbour porpoises. The maximum distances from the turbine at which either species started to experience any LSR were larger during winter (corresponding to a period with lower levels of ambient noise). The implication is that animals experience variable listening spaces around tidal energy generators (or other anthropogenic sources) depending on the time of year.

USA: Fouda *et al.* (2018) reported on the effects of elevated ambient noise levels, mainly driven by commercial shipping, on social whistle calls produced by bottlenose dolphins (*T. truncatus*) in the western North Atlantic. Data were collected using a bottom-mounted SM3M recorder (Wildlife Acoustics) sampling at 48 kHz during July–September 2016, located approximately 30 km offshore of Maryland, USA. A total of 200 high-quality whistles from 16 encounters, in the frequency range 2.93–23.83 kHz, were available for in-depth analysis, using a range of characteristics (e.g. whistle length, start and end frequency, whistle contour shape [incl. number of inflection points], and presence of harmonics). Ambient broadband noise levels ranged between 108.1–134.2 dB re 1 µPa, and significantly affected numerous whistle characteristics. Broadly, dolphins were observed to both increase the frequency and simplify the structure of their social whistles, presumably to reduce the risk of information loss to the call recipient due to masking by ambient (anthropogenic) noise. This response has also been noted in other species of marine mammals (Lesage *et al.*, 1999) as well as other terrestrial species. Importantly, dolphins adjusted their whistles even when exposed to ambient noise at frequencies lower than their whistle calls, suggesting that signals were becoming difficult to distinguish against increasing levels of low-frequency ambient noise (informational masking; Clark *et al.*, 2009). The authors raise concerns that such noise emissions are becoming increasingly common in urbanised coastal waters, with potential deleterious effects on dolphin communication and group cohesion.

A study by Cholewiak *et al.* (2018) investigated the degree to which anthropogenic noise affected the ability of marine mammals to communicate over large distances („communication space" or CS, following Clark *et al.* (2009)). Data on vocalisations of fin whale, minke whale, humpback and North Atlantic right whale, and ambient noise were collected using arrays of autonomous passive acoustic recorders deployed for up to three months within the Stellwagen Bank National

Marine Sanctuary (SBNMS) off Massachusetts, USA (Hatch *et al.*, 2008; 2012). These data were integrated into an agent-based modelling approach to determine the available communication space for individual animals across a 24-hour period in ten minute intervals. The models also made use of available movement and position data for three categories of vessels: large commercial/passenger/military vessels (mainly using AIS data), fishing vessels (using VMS data) and whale-watching vessels (modelled based on data collected from such vessels at the appropriate time of year). Where empirical data were available, received levels (RLs) were calculated in one-third octave bands for each known vessel using the closest point of approach (CPA) of the vessel to an acoustic receiver. Once transmission loss was accounted for, both maximum potential CS and actual available CS were calculated for typical calls produced by the four baleen whale species. The results indicated that ambient noise conditions as observed in the SBNMS resulted in significant reductions in communication range in excess of 50%, particularly for fin, minke and humpback whales, while North Atlantic right whale calls were less affected. Large commercial/passenger/military vessels contributed most to this CS reduction, which ranged as high as 99% for fin whales under the noisiest conditions, likely due to the poor propagation conditions for low-frequency sounds encountered near Stellwagen Bank. The study highlighted the potential risk of anthropogenic noise pollution on the ability of these baleen whales to successfully communicate amongst themselves, with unknown but potentially significant consequences for these species.

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2 ToR B. Review and update information on the ecological role of marine mammals

The objective of this ToR is to provide an overview of the ecological role of marine mammals, e.g. influence on ecosystem structure and function, transfer of energy and matter as well as parasites in marine foodwebs. This section thus widens the scope of the ecological role of marine mammals and complements the review of indirect impacts of seals on fisheries presented in the previous WGMME report, covering diet, prey consumption, impact on fish stocks and competition for food (ICES, 2018). Besides the related subject in the report from 2018 (ICES, 2018), WGMME has previously reviewed interactions between grey seals and other marine mammal species (ICES, 2016), multispecies models incorporating marine mammals (ICES, 2015), marine mammal diet (ICES, 2004; ICES, 2006) and ecosystem impacts of seal culling programmes (ICES, 2004). Given the limited resources available to produce this review, studies may have been overlooked.

2.1 Ecological role of marine mammals

Marine mammals can play an important ecological role in marine ecosystems. Being a diverse group of dynamic predators, marine mammals feed at different trophic levels and are capable of transfer of energy and nutrients as well as regulation of abundance of other species and ecosystem structuring effects (Bowen, 1997; Trites, 2001; Morissette *et al.*, 2006). However, marine foodwebs are often complex, and predation on one level of the foodweb may have effects also on other levels e.g. via predation or competition. Thus, to obtain a complete understanding of the ecological role of marine mammals is a challenging task, and the available literature looking at the ecological role of marine mammals is limited (Estes *et al.*, 2016b). In general, a perfect and absolute understanding of the ecological roles of marine mammal species in a complete ecosystem context is unlikely. Instead, contexts and models reduced to essentials need to be used to progress the knowledge. Caution needs to be taken, however, not to oversimplifying foodweb interactions leading to management measures with unforeseen and perhaps even harmful impacts.

Typically, studies of the ecological role of marine mammals have been focused on their consumption of prey and its consequences within the specific foodweb. However, marine mammals are sometimes also prey (Weller, 2009) and, consequently can have impact on their predators (bottom–up) as well as on lower trophic levels (top–down). In addition to being prey of other predators or consumers of prey, marine mammals could possibly also have effect on the ecosystem through impact on behaviour, life history and morphology of prey populations (Power and Gregoire, 1978; Heithaus *et al.*, 2008; Wirsing *et al.*, 2008; Benoit *et al.*, 2010; Kiszka *et al.*, 2015). Other non-feeding effects constitute increasing the abundance of prey for other species, e.g. seabirds (Dill *et al.*, 2003; Anderson and Lovvorn, 2008), facilitating transport of nutrients (Doughty *et al.*, 2016), habitat modifications, i.e. ecosystem engineering (Roman *et al.*, 2014), spreading of parasites (McClelland, 2002; Haarder *et al.*, 2014) and even transmission of diseases and invasive species (Hammerschlag *et al.*, 2019). Ecosystem functions together with ecosystem services associated with marine mammals and other aquatic predators have been reviewed in a recent paper by Hammerschlag *et al.* (2019).

2.2 Marine mammal influence on ecosystem structure and function

Marine mammals play an important part in the functioning of marine ecosystems and can have influence on several groups and levels in the foodweb, both directly by prey consumption and indirectly by other foodweb interactions e.g. competition, cascading effects on lower trophic levels and manipulation of the environment by ecosystem engineering (Swain and Sinclair, 2000; Frank *et al.*, 2005; Baum and Worm, 2009; Roman *et al.*, 2014; Estes *et al.*, 2016b; Aarts *et al.*, 2019). As foodweb dynamics in marine ecosystems are complicated in general, there are more than just direct predator–prey interactions. Marine mammals can consume prey species which may, in turn, have impact on other species, e.g. by being a food resource, a predator as well as a competitor of other species. For example, if a prey species has negative impact on other species, which can be of commercial interest, the marine mammal predation may even be beneficial for the abundance of a commercial species (Punt and Butterworth, 1995; Li *et al.*, 2010; Baudron *et al.*, 2019).

Marine mammals have the potential to create habitat from carcasses of their prey or the marine mammals themselves. It is likely that dead marine mammals provide an immense amount of energy to predatory species in both marine and terrestrial ecosystems. The role of scavengers in trophic models is generally overlooked, and the value of whale and seal carcasses in both the marine and terrestrial ecosystems is likely underestimated. Scavenging occurs in virtually all ecosystems containing carnivores and is an important energy transfer pathway that can impact ecosystem structure and function (Fallows *et al.*, 2013). The ecological significance of this has mostly been considered in terrestrial ecosystems. In marine systems, marine mammal carcasses provide foraging opportunities for a range of scavengers, linking a number of different ecosystems by transferring energy to aerial, terrestrial and aquatic scavengers, across a range of taxonomic groups (Katona and Whitehead, 1988; Fallows *et al.*, 2013; Lewis and Lafferty, 2014; Gallagher *et al.*, 2018; Laidre *et al.*, 2018; Lambertucci Sergio *et al.*, 2018). Numerous studies have looked at the biodiversity and ecological succession patterns of benthic scavengers on sinking marine mammal carcasses, including agnathans, chondrichthyan and teleost fish, numerous species of invertebrates, including crustaceans, echinoderms and molluscs (Jones *et al.*, 1998; Smith *et al.*, 2015; Alfaro-Lucas *et al.*, 2018). With an increase in decomposition, the resulting microhabitats, along with the surrounding sediments support foodwebs with greater trophic complexity, dominated by higher consumers mainly relying on marine mammal and/or photosynthesis-derived organic matter, a type of foodweb commonly reported in small whale, wood and kelp falls (Alfaro-Lucas *et al.*, 2018).

Marine mammal species can also alter benthic habitats and release substantial amounts of nutrients by bioturbation, i.e. affecting the benthic topography and enhancing food availability for other species (Nelson and Johnson, 1987; Alter *et al.*, 2007). In addition, marine mammals can engineer their ecosystems indirectly through trophic interactions, in which the marine mammals have impact on other species that, in their turn, modify the habitat (Crump *et al.*, 2010; Estes *et al.*, 2016a).

2.3 Marine mammal influence on energy transfer

Marine mammals, large whales in particular, are considered important in the transfer of nutrients from deep water to surface water by releasing faecal plumes at the surface, after feeding in deeper waters. The animals sequester carbon to the deep sea; whale falls are thought to transfer an estimated 190 000 tons of carbon per year from the atmosphere to deeper waters, and declining top predator populations, especially large species such as baleen whales, lower the potential for marine ecosystems to retain carbon (Pershing *et al.*, 2010). Marine mammals also influence biogeochemical cycles by releasing nutrients including nitrogen in urine and faeces (Lavery *et al.*, 2010; Nicol *et al.*, 2010; Roman and McCarthy, 2010). Similarly, because of migration patterns, marine mammals may also move nutrients from highly productive feeding grounds, to (otherwise) relatively unproductive breeding grounds. Nutrients may also be transported from offshore environments to coastal as well as terrestrial environments, having effect on terrestrial plants and animals (Doughty *et al.*, 2016; McLoughlin *et al.*, 2016), but see also Moss (2017).

2.4 Marine mammal transfer of parasites in foodwebs

Knowledge of marine mammal parasite diversity and prevalence data as well as information about their transmission pathways may be helpful to assess parasite effects on host and foodweb ecology (Poulin *et al.*, 2016). Marine mammal parasites can be used as bio-indicators of host habitat use and population dynamics (Balbuena and Raga, 1994; Aznar *et al.*, 1995) as well as their diet (Sinisalo *et al*. 2006; Marcogliese, 2002). Parasites are used as markers for marine mammal health status and exposure to pollutants (Pascual and Abollo, 2005; Aznar *et al*., 2005; Siebert *et al*., 1999; Lehnert *et al*., 2014). Monitoring parasite infections in systematic investigations of parasite fauna and its impact on marine mammal and ecosystem health can provide important long-term data. Minimally invasive tools like analysing blood or faeces to investigate live animals and complement post-mortem investigations have been used in studies on odontocetes (Kleinertz *et al*., 2014) as well as baleen whales (Hermosilla *et al*., 2015) and pinnipeds (Ulrich *et al*., 2016; Hermosilla *et al*., 2018). Understanding foodweb and ecosystem interactions can be furthered by including parasitology data (Lafferty *et al*., 2008d; Poulin *et al*., 2016), although information about biodiversity and life history of many marine parasites is missing (Poulin, 2004). Because cestodes, nematodes and acanthocephalans mostly rely on trophic transmission, they are useful indicators of persistent foodweb interactions (Marcogliese, 2002). Many species have heterogenous transmission pathways including multiple invertebrate and vertebrate intermediate or paratenic hosts that are difficult to identify. Trophically transmitted helminth parasites use predator–prey links for their own transfer from intermediate prey hosts, in which they occur as larval stages, to predatory final hosts, in which they reach maturity (Poulin and Leung, 2011). Poulin and Leung (2011) investigated how size and trophic level of fish influence the probability of being used as intermediate host and found that the proportion of larval helminth taxa in fish shorter than 20 cm was twice as high as that for fish over 100 cm in length concluding that trophic level was no reliable predictor of the proportion of helminth species occurring as larvae in a fish (Poulin and Leung, 2011).

2.5 Inclusion of parasites in foodweb assessments

Although parasites can have ample effects on ecosystems, endoparasites were included in foodwebs only recently (Lafferty *et al*., 2008c). The first studies integrating parasites into foodwebs observed increases in species richness, number of links, and food chain length. Further studies also found that their addition increased connectedness and nestedness (Lafferty *et al*., 2006). Being embedded in foodwebs makes parasites sensitive to changes in the environment. In particular, fishing and environmental disturbance, by reducing fish populations, may reduce parasite populations. Indirect evidence indicated a decrease in parasites in commercially fished species over the past three decades (Lafferty *et al*., 2008a,b). While parasites were initially neglected because their relative biomass was considered low compared with that of other trophic groups, nowadays there is evidence that parasite-mediated effects are important for energy flow and production in ecosystems (Hudson *et al*., 2006). In a study quantifying the biomass of parasites in three Pacific estuaries, it was shown that parasite biomass was substantial and sometimes even exceeding that of top predators (Kuris *et al*., 2008). Trophically transmitted parasites consumed more biomass than did other parasitic groups implying the important role for infectious processes in these areas.

Employing network analysis to look at specific changes parasites induce when included in foodwebs showed that results reflect the consumer character of parasites and the close physical proximity to their hosts, as well as their complex life cycles and small body size (Dunne *et al*., 2013). Another study reviewed how network analysis can be useful to gain knowledge of parasite ecology, host parasite interactions and foodweb complexity (Poulin, 2010).

Theoretically, a correlation between host and parasite population sizes could be expected. A study analysing intestinal parasites (acanthocephalans *Corynosoma semerme*, *C. strumosum* and *C. magdaleni*) in ringed seals (*Pusa hispida*) from the Bothnian Bay over 20 years found that although seal numbers during this period first decreased markedly and then increased steadily, no change in the mean abundance of any of the three acanthocephalan species over time was found, nor was there a relationship between parasite abundance and seal numbers in the corresponding year (Valtonen *et al*., 2004). This stability also persisted although a paratenic fish host particularly important for *C. strumosum* had gradually disappeared at the same time from the bay due to decreasing salinity.

Parasites are also known to influence the behaviour of their hosts (Sato *et al*. 2012; Lafferty *et al*., 2008d; Poulin *et al*., 2016) and therefore have to be taken into account when looking at behavioural drivers for ecological roles of fish and mammals in the marine ecosystem (Kiszka *et al*., 2015).

In conclusion, parasites are useful as indicators for many aspects of intermediate and final host ecology, can inform about trophic interactions and foodweb structure and parasite studies can provide scientifically organized data on which to base decisions for management and conservation.

2.6 Review on the occurrence of trematode gastric parasites

Trematodes belonging to the family Opisthorchiidae play an important role as fish-borne parasitic zoonoses in humans and can cause serious disease such as cholangitis and pancreatitis in certain areas of the world, especially in Asia where fish are often consumed raw (EFSA, 2010). Opisthorchiidae infections can affect the hepatobiliary system of humans after ingestion of raw or undercooked fish containing infective metacercariae (Näreaho *et al*., 2017).

Recently, emerging infections with the trematode *Pseudamphistomum truncatum* have been found in Baltic grey seals (Neimanis *et al*., 2016) from Sweden. The endoparasitic fluke belonging to the Opisthorchiidae family is a generalist parasite that infects the biliary tract of various fish-eating mammals and has zoonotic potential, also for humans. It has been described in e.g. Caspian seals (*Pusa caspisca)* (Heckmann *et al*., 2014), in otters (*Lutra lutra)*, mink (*Mustela vison*) (Simpson *et al*., 2005) and fox (Saeed *et al*., 2006) in areas around Europe ranging from the Gulf of Finland to Denmark, Ireland and the UK (Sherrard-Smith *et al*., 2009).

The life cycle of opisthorchiid trematodes is incompletely understood but involving several intermediate hosts, usually including freshwater gastropods and cyprinid fish. Roach (*Rutilus rutilus*) are suspected to play a key role as metacercariae of *P. truncatum* have been demonstrated in this fish species in Germany, Denmark and Ireland (Näreaho *et al*., 2017; Skov *et al*., 2008; Hawkins *et al*., 2010).

Approximately 12% of Baltic grey seals investigated between 2002 and 2013 were shown to have hepatobiliary trematode infections (Neimanis *et al*., 2016) and their diet contained significantly larger proportions of cyprinids than during an earlier period, coinciding with an increase of trematode parasitism in the seal host population. Although most infections were reported to be mild, *P. truncatum* can cause severe hepatobiliary disease and mortality in the investigated seals (Neimanis *et al*., 2016,). Salinity, changes in cyprinid abundance, viability of gastropod intermediate hosts as well as changing immune competence and food preferences of seals could be additional factors influencing *P. truncatum* prevalence. Neimanis *et al*. (2016) state that seals prove to be sentinels of ecosystem change and that *P. truncatum* occurrence and identification of potential intermediate fish hosts warrants further investigations because of human health concerns.

Although there have been reports of trematode infections from the 1988/89 PDV epidemic reporting *Cryptocotyle lingua* and *Phagicola septentrionalis* in the intestinal tract of harbour seals

(Strauss *et al.*, 1991), since then no trematodes have been recorded in seals from the German North Sea and Baltic Sea (Siebert *et al.*, 2007; Lehnert *et al.*, 2007).

In cetaceans, the generalist trematode *Pholeter gastrophilus* occurs in the gastro-intestinal tract embedded in the mucosa (Aznar *et al.*, 2006; Lehnert *et al.*, 2005). This gastric digenean has been reported from multiple cetacean species worldwide (Fraija-Fernandez *et al.*, 2017). The parasite seems to prefer coastal species (Gibson *et al.*, 1998) but has also been found in oceanic (Fernandez *et al.*, 2003) and freshwater species (Zam *et al.*, 1970). Little is known about its life cycle, but it is assumed that molluscs and fish act as first and second intermediate hosts. *P. gastrophilus* burrows into the stomach wall in submucosal fibrotic nodules that can contain from one to several worms and have a narrow duct that opens into the stomach lumen to release eggs (Gibson *et al.*, 1998). The trematode was reported to be accompanied by mild granulomatous inflammation (Lehnert *et al.*, 2005) but also irritation, haemorrhages and fibrosis of the stomach mucosa (Gibson *et al.*, 1998) and seemed to have no serious consequences for the host (Siebert *et al.*, 2001).

Adult digeneans belonging to the Brachycladiidae family, infect mainly hepatic and pancreatic ducts of marine mammals worldwide (Raga, 1994). 35 of ca. 42 described species are found in cetaceans with additional species infecting pinnipeds (Andersen-Ranberg *et al.*, 2018) and otters (Aznar *et al.*, 2001; Dailey, 2007). Nothing is known about the life history of the group as no intermediate or paratenic hosts have been identified. They express strong species-specific traits with some species infecting only single odontocete species (*Campula oblonga* in harbour porpoises) (Gibson *et al.*, 1998) and others occurring in a wide range of sympatric cetaceans (*Lecithodesmus goliath*) (Deliamure, 1961) or even in pinnipeds as well as cetaceans (*Orthosplanchus arcticus*) (Raga and Balbuena, 1993). Mateu *et al.* (2011) investigated the liver fluke infection patterns of a group of dolphin species in the Mediterranean to better understand host specificity of these trematodes, necessarily confronted with the opportunistic sampling drawbacks encountered in marine mammal parasitology research.

The trematode *Campula oblonga* is reported to induce light pathologies like duct obstructions and fibrosis (Raga and Balbuena, 1992), but also cholangitis, chronic pancreatitis, chronic hepatocholangitis and portal and periportal fibrosis of the liver and hyperplasia of the bile-ducts (Gibson *et al.*, 1998; Siebert *et al.*, 2001). In porpoises from German waters and around Iceland infections in the pancreas were also found, causing periductular fibrosis, eosinophilic inflammation of pancreatic ducts and pancreatitis (Siebert *et al.*, 2001; Lehnert *et al.*, 2014).

2.7 The role of fish in the life cycles of respiratory parasites of marine mammals

Nematodes associated with the respiratory tract of marine mammals often cause secondary bacterial infections, inflammatory lesions, and mortality (Measures *et al.*, 2001; Siebert *et al.*, 2001). One particularly pathogenic group of parasites are nematodes belonging to the superfamily Metastrongyloidea. They are usually found in the bronchi, parenchyma, or blood vessels associated with the lungs and cause pneumonia, impede diving ability and are hence an important factor regarding the viability of wild populations (Measures and Gosselin, 1994; Gulland and Beckmen, 1997; Jepson *et al.*, 2000; Siebert *et al.*, 2001; 2007). Parasite-induced bronchopneumonia is a frequent cause of mortality in odontocetes (Gibson *et al.*, 1998; Wünschmann *et al.*, 2001) and pinnipeds (Dailey, 1970; Onderka, 1989; Gosselin *et al.*, 1998). Lesions caused by nematodes in the lung tissue (Wünschmann *et al.*, 1999) have been discussed as possible entrance for infective agents and may facilitate infections by viruses, bacteria and fungi.

In harbour seals, two species of lungworms occur that belong to two different families (Crenosomatidae and Filaroididae) that have adapted to different niches in the respiratory tract and members of which also parasitise terrestrial mammals like carnivores and insectivores. In harbour porpoises, four species belonging to the Pseudaliidae, a family extinct in terrestrial mammals, are inhabiting different niches in the circulatory system, bronchi and parenchyma and the cranial and auditory sinuses (Lehnert *et al.*, 2005). Lungworms infecting marine mammals derived from terrestrial ancestors, and are assumed to have followed their hosts to the marine environment. Although a period of coevolution between marine mammals and their lung parasites is assumed (Anderson, 1982) little is known about the ecology and transmission of lungworms in the marine environment (Dailey, 1970; Lehnert *et al.*, 2010). There is evidence of prey intermediate hosts (Dailey, 1970; Houde *et al.*, 2003; Lehnert *et al.*, 2010) of pinnipeds and cetaceans (Dailey, 1970; Houde *et al.*, 2003; Lehnert *et al.*, 2010), but other studies have indicated that direct infections of *Halocercus* species are possible in bottlenose dolphins (*Tursiops truncatus*) (Dailey *et al.*, 1991; Fauquier *et al.*, 2009) and Australian common dolphins (*Delphinus delphis*) (Tomo *et al.*, 2010). In experimental studies on marine metastrongyloids, American plaice (*Hippoglossoides platessoides*) and Arctic sculpins (*Myoxocephalus scorpioides*) were reported to act as potential intermediate hosts for *Pharurus pallasii* in beluga whales (*Delphinapterus leucas*) in Canada (Houde *et al.*, 2003). *Girella nigricans*, a coprophagic fish, is a suitable intermediate host in the transmission of *P. decorus* to California sea lions (Dailey, 1970) and American plaice were reported to act as a potential intermediate host for *O. circumlitus* in ringed seals (*Pusa hispida*) (Bergeron *et al.*, 1997). Another study found infective third-stage larvae of *P. inflexus*, *T. convolutus*, *S. minor* and *P. gymnurus*, lungworm species of harbour porpoise and harbour seal, respectively, in the intestinal wall of several flatfish species (mostly dab (*Limanda limanda*) and plaice (*Pleuronectes platessa*)) (Lehnert *et al.*, 2010). The study also reported on experimental infections of *O. circumlitus* to turbot (*Psetta maxima*), indicating that benthic fish species in the prey spectrum of cetaceans and pinnipeds play a role in the transmission of metastrongyloid nematodes (Lehnert *et al.*, 2010). However, in one report on mother-to-calf infections with *Halocercus lagenorhynchi* in bottlenose dolphins, the authors suggested that transplacental infection may take place in some pseudaliid species infecting cetaceans (Dailey *et al.*, 1991). In this previous study, four dolphin calves displayed multiple mature nematodes in the lung tissue. Reckendorf *et al.* (2018) reported for the first time on pseudaliid nematode infections in the lungs of two neonatal orcas that were probably infected by their lactating mothers. Recent studies support that prenatal infections of lung nematodes in cetaceans are probable. Neonatal bottlenose dolphins investigated in South Florida were found to have the highest prevalence and severity of *H. lagenorhynchi* infections compared to juveniles and adults (Fauquier *et al.*, 2009) and lungworm infections (*H. lagenorhynchi*, *Pharurus alatus*, *Stenurus ovatus*) in dependent calves of Australian common dolphins and bottlenose dolphins were found to be more prevalent than in juveniles or adults (Tomo *et al.*, 2010). However, the likelihood of pseudaliid lungworms having differing, species-specific life cycles or being able to use both direct and indirect transmission pathways requires further study (Reckendorf *et al.*, 2018).

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

3.1 Background

This ToR aims to identify and, whenever possible, report on aspects of marine mammal fishery interactions not currently covered by WGBYC. In consultation between the chairs of WGBYC and WGMME, and subsequent discussion in initial plenary session at the 2019 meeting of WGMME, the following aspects were highlighted:

i. Seal interactions with fisheries (e.g. in which fisheries and by whom is it reported; how many seals are bycaught in fisheries);

ii. Reviewing other sources of cetacean bycatch data, i.e. those not being used in current bycatch assessments: the nature and utility of information available on cetacean bycatch, by area/country, gear, species. It is noted that WGBYC compiles data from National Reports from the MS related to EC Regulation 812/2014 and that in the last two years ICES has issued data calls for information on fishing effort, monitoring/sampling effort and bycatch numbers obtained from other monitoring programs, including the fisheries Data Collection Framework (DCF, now EU-MAP). However, at present, WGBYC does not routinely make use of information from strandings, voluntary recording schemes, interview surveys, etc.;

iii. Identify potential risk areas (i.e. species-area-fishery combinations associated with a high risk of negative impacts on marine mammal populations due to fishery bycatch mortality). Given the limited coverage and implementation of Regulation 812/2004, a comprehensive evaluation of bycatch risk is not possible using dedicated on-board observer data alone. Relevant information is however expected to emerge from other data sources examined in the course of work under points (i) and (ii) above. This work should also allow us to identify information gaps;

iv. Revise existing thresholds for bycatch: WGMME reviewed this topic in 2013 and 2014, describing various approaches used to set safe limits for bycatch mortality in marine mammal populations. Here we will review recent work on the topic.

It was noted that, due to time constraints, some of these topics would not be covered fully this year and WGMME therefore proposes to continue work on these and/or related topics in 2020, with the precise content of the ToR to be decided following discussion with WGBYC.

Several topics under this ToR have been addressed previously by various ICES Expert Groups and/or by organisations such as ASCOBANS, the International Whaling Commission (IWC) and the North Atlantic Marine Mammal Commission (NAMMCO). The IWC recently instigated both Strandings and Bycatch Mitigation Initiatives, each involving a Steering/Working Group, an Expert Panel and a coordinator. We therefore aim to summarise previous work on each topic at the start of the relevant section.

3.2 (C i) Seal-fishery interactions: seal bycatch

3.2.1 Rationale

Indirect seal-fisheries interactions (e.g. related to overlap between seal diet and fishery catches and to transmission of parasites) were highlighted in ICES reports at least since the 1970s (e.g. Sergeant, 1970; Söderberg, 1971; Kapel, 1973). Aspects of this topic have previously been reviewed by WGMME on several occasions (e.g. ICES, 2004; 2006; 2015a). WGSAM has used ecosystem models to quantify the impacts of seals or cetaceans on fish stocks (e.g. ICES, 2018).

The first ICES reports on fishery bycatch mortality of seals appeared in the mid-1980s (e.g. Sjöåsen, 1986). Prior to the formation of SGBYC (2010), which became WGBYC in 2011, fishery bycatch mortality in marine mammals was reported by SGSSC (1991–1996) SGBS/WGBS (Baltic seals, 1989–1990), WGHARP (harp seals and hooded seals, ongoing since 1990), WGMMPD (1998–1999), WGMMPH (2001–2002) and WGMME (2003–2007).

A historical data gap has arisen because WGBYC focused primarily on cetacean bycatch, in particular collating and analysing information provided by Member States (MS) in response to the requirements of Regulation 812/2004. There appears to have been no systematic reporting of seal bycatches to ICES (except for harp and hooded seals) since 2007.

Grey seal populations have increased in almost all areas of the Northeast Atlantic in recent decades, while harbour seal populations show varying trends, including increases in some areas (e.g. the Wadden Sea, Kattegat, SW Baltic and Baltic Proper). Where seal numbers have increased, this has led to an increase in interactions between seals and fisheries (e.g. Olsen, 2018). These interactions include, among others, seals predating on fish in fishing nets and at aquaculture facilities, and seal bycatch (e.g. Cosgrove *et al.*, 2013; see also ICES, 2017; 2018).

Some MS have reported information on bycatch of seals (and other protected species) to WGBYC as part of their reports under Regulation 812/2004 or within other reports. Thus, in 2015, WGBYC reported that five MS had provided information on bycatch of grey and harbour seals (ICES, 2015b). It should be highlighted that Regulation 812/2004 does not require monitoring for species other than small cetaceans and therefore seal bycatch reporting was on a voluntary basis, using dedicated (marine mammal) observers deployed to record cetacean bycatch for area/gear combinations where such monitoring is mandated under 812/2004.

With the move from dedicated monitoring of bycatch (under 812/2004) to its incorporation within fisheries monitoring under EU MAP (formerly DCF), fishery observers will be required to record bycatch of cetaceans, seals and other protected species, and these data will be submitted to ICES (see Section C ii for further detail).

In this section, we summarise information on seal bycatches that is currently recorded / available, on a country by country basis.

Belgium: Monitoring of seal bycatch is based on collection and investigation of carcasses washed ashore. Bycatch has been observed in the beach set-net fishery but this concerns at the most a couple of animals per year. Increasing numbers of harbour seals are observed hauling-out, especially in the port of Nieuwpoort but numbers (5–15) remain low compared with haul-out sites or colonies in neighbouring countries. No births have taken place. In 2017, at least four animals were injured by fishing hooks (Figure 1) and at least one animal died due to ingestion of fishing hooks (Haelters *et al.*, 2018), while in 2018 at least three animals were injured by hooks at Nieuwpoort. In 2018, the number of dead and dying seals recorded by the Belgian strandings monitoring network was the highest ever recorded, totalling 43 animals (17 grey seals, 12 harbour seals and 14 unidentified seals). One of the grey seals had died due to a rope around its neck (RBINS data, unpublished).

Figure 1. A harbour seal injured by fishing hooks, Nieuwpoort, Belgium, 4 December 2017 (Photo: Linda Vanthournout). (The animal is still alive at the time of writing).

Finland: Increasing numbers of grey seals and ringed seals in the Baltic Sea have resulted in increasing conflict between seals and fisheries. Vanhatalo *et al.* (2014) estimated that 2000 grey seals were dying per year by trap and gillnets in Finland, Sweden, and Estonia, based on analysis of interview data (from 2012) using a hierarchical Bayesian model. No estimates are available for the ringed seals.

Germany: In the framework of two dedicated stranding networks, marine mammal carcasses are regularly reported and collected in Germany (federal states of Schleswig-Holstein SH (North Sea and southwestern Baltic Sea) and Mecklenburg-Western Pomerania MWP (Baltic Sea)). In SH, beaches are patrolled twice a day by seal rangers as well as trained volunteers since 1990 (Siebert *et al.*, 2001). Additionally, strandings are also reported by tourists. Carcasses are then collected for further investigations. Only marine mammals that were identified as bycaught and handed over by fishermen are classified as 'bycatch'; bycatch determined among strandings, based on pathological investigations following necropsies, are classified 'suspected bycatch'.

In MWP (Baltic Sea), between 2016 and 2017, no seal bycatch has been handed over directly by fishermen. In fall 2017, between 10 September–4 December 2017, an unusual large number of 23 male grey seals was recorded stranded, all within the same area southeast of the island of Rügen (Westphal *et al.,* 2018). Following necropsies, 15 of those were categorised as suspected bycatch (Dähne *et al.,* 2019).

In SH (North and Baltic Sea), five bycatches of harbour seals were reported between 1995–2017 (three bycatches in the North and two in the Baltic Sea).

In Lower Saxony (southern German Bight), strandings are partly reported but not systematically investigated. Therefore, the impact of fisheries on seals (and harbour porpoises) for those waters is unknown.

Ireland: Evidence from dedicated observer projects indicates a low risk of seal bycatch in gillnet fisheries off the west and southwest coasts of Ireland and high risk in large mesh tanglenets targeting crawfish in the same area (Cosgrove *et al.,* 2016). Monitoring of these fisheries is not required under Regulation 814/2004 so bycatch data have been obtained via dedicated short-term projects and coverage has therefore been patchy. It is also difficult to obtain accurate fishing effort data (Cosgrove *et al.,* 2016).

Results from marking and tracking of individual grey seals, relating seal movements to the distribution of fishing effort on the Celtic Shelf and in the Irish Sea, indicate overlap with inshore passive gear fisheries but not with fisheries using active gears (i.e. demersal and pelagic trawlers targeting cod, hake, haddock, pollack, herring, and mackerel) (Cronin *et al.,* 2016).

Netherlands: Seal strandings in the Netherlands can be reported by any member of the public to a public database ¨Waarneming.nl¨ (Brasseur, 2017). Over 15 000 records of seal strandings are now available covering a period going back several decades. Although reporting effort and cause of death are not recorded, the database is a useful tool to identify areas of high mortality. If put together with abundance data, such information could help identify areas where seal-fishery interactions are an issue.

Scotland: Beach-cast seal mortalities are routinely collated and investigated by pathologists at Scottish Marine Animal Stranding Scheme (SMASS). Results are published quarterly and summarised in annual reports. Pathological indications of bycatch are investigated in the subset of strandings taken to necropsy but incidence rate is very low. There is no reporting of seal bycatch from vessels to SMASS.

Sweden: Seal bycatch is not monitored on a regular basis. Stranded and bycaught animals are reported on a voluntary basis and a decision is made as to whether carcasses are sent to the Swedish Museum of Natural history for necropsy. Figure 2 illustrates the number of bycaught grey seals from the Baltic, by age group, necropsied during 2002–2017 (Bäcklin, 2017, pers. comm.).

A study by Lunneryd *et al.* (2004) based on interviews with fishermen, estimated that 462 (95% CI 360–575) grey seals, 52 (95% CI 34–70 ringed seals and 461 (95% CI 333–506) harbour seals were bycaught in Sweden in 2001. Vanhatalo *et al.* (2014) collected interview data in Sweden, Finland and Estonia in 2012 and used a hierarchical Bayesian model to estimate total bycatch of grey seals in the Baltic. The posterior median and mean for total grey seal bycatch by trap and gillnet fisheries were 1550 and 1880, respectively (90% confidence limits 1240 to 2860). The authors noted that estimates based on data collected during interviews are likely an underrepresentation of the true number of bycaught seals.

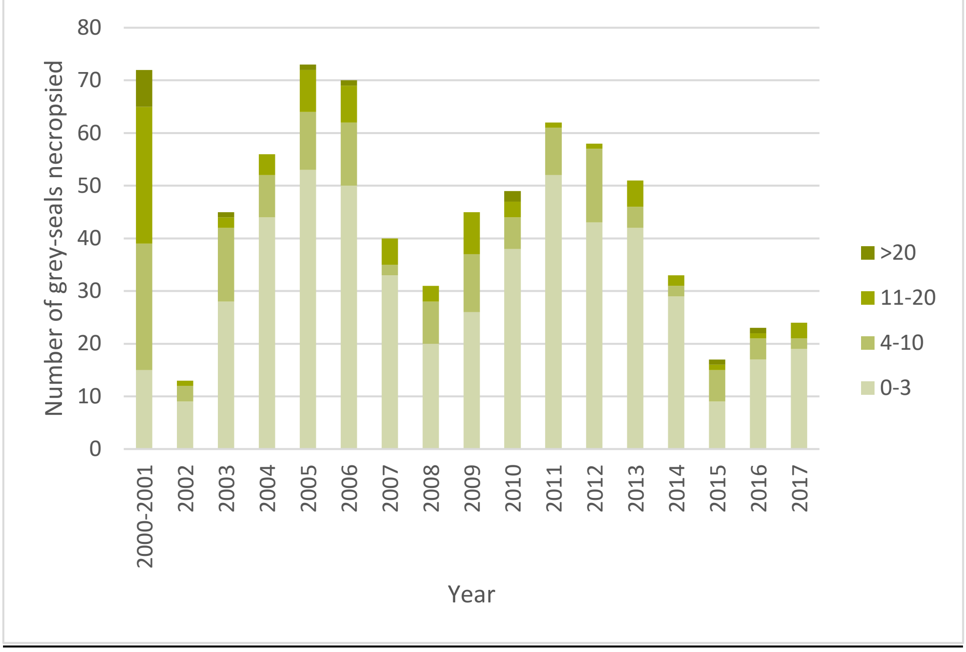

Figure 2. Bycaught grey seals, by age group, necropsied at the Swedish Museum of Natural History (Bäcklin, 2017, pers. comm.).

3.2.2 Recommendation

There is a need to ensure that fishery bycatch of seals in the ICES area is monitored and reported. While the natural ICES home for compilation and analysis of the resulting information is probably WGBYC and WGHARP, WGMME offers to assist as required.

3.3 (C ii) Other bycatch data

3.3.1 Background and Rationale

Until recently, WGBYC has worked almost entirely with data collected under Regulation (EC) No. 812/2004, which (normally) consists of reports of cetacean bycatches from on-board deployment of dedicated cetacean observers. As ICES has stated on several occasions, the data provided by MS as part of the requirements of Regulation 812/2004 have proven insufficient to "allow robust assessment" of the level of cetacean bycatch. Known issues include exclusion of many relevant fleets, partial implementation, low sampling intensity and non-random selection of boats for monitoring.

Regulation 812/2004 is due to be repealed following Commission Implementing Decision (EU) 2016/1251 (EU MAP), which specifies new requirements for documenting incidental bycatch of all birds, mammals and reptiles and fish protected under EU legislation and international agreements in all types of fisheries. The monitoring function of Regulation 812/2004 is being transferred to the DCF, while its mitigation functions are being subsumed under the new Technical Measures framework (e.g. see ICES, 2016).

ICES WGBYC therefore anticipates making use of data from DCF fishery observers in order to assess protected species bycatch and ICES has issued two data calls *to improve consistency and completeness in the reporting of bycatch data at a regional scale.*

In most cases, MS have reported to the Commission that they will start the collection of bycatch data and/or other information to assess the impact of fisheries on the ecosystem through pilot studies, while others reported that they will incorporate bycatch reporting into the tasks of fisheries observers. The latter raises issues as to whether a common sampling design for fisheries and bycatch monitoring would provide data suitable to allow robust bycatch estimates. There are also questions related to observer training needs and the efficacy of combining bycatch monitoring with other observer duties.

Even if these issues of sampling design and data quality are solved, there is still likely to be a time delay before the sampling is fully implemented and all MS report the information. There is therefore, perhaps more than ever, a need to explore other sources of data that can help evaluate the occurrence and amount of bycatch mortality and/or help identify species/area/fishery combinations that may be associated with high bycatch mortality.

For some of those cases, available data may never be sufficient to provide accurate, precise and robust bycatch estimates, as indeed may be the case for abundance surveys, thus leading to high statistical uncertainty about the bycatch rate. However, where population size is small and known minimum number of bycatches is relatively large, it should be evident that precautionary management action is needed to mitigate the threat from bycatch. A case in point is the Iberian harbour porpoise, a genetically distinct small population which appears to suffer high and likely unsustainable levels of bycatch mortality but for which data collected under 812/2004 are inadequate to make a robust assessment (North Atlantic Marine Mammal Commission and the Norwegian Institute for Marine Research 2019).

In its 2011 report, WGBYC briefly reviewed the potential utility of other sources of information such as strandings (ICES, 2011). From the above, it therefore seems timely to revaluate the utility of other sources of data on marine mammal bycatch, e.g. strandings monitoring, voluntary reporting schemes such as electronic logbooks, interview surveys, REM (Remote Electronic Monitoring systems with Closed-Circuit Television, CCTV) and to consider how they could be integrated with observer data. There is currently no comprehensive review of bycatch information derived from such sources for EU waters.

3.3.2 Strandings

Of all possible additional sources of data on bycatch mortality, strandings (marine mammals that appear on the coasts dead or alive) probably has the potential to offer the most useful insights into the scale of the bycatch issue. Nevertheless, while the usefulness of strandings data has been proven in studies of the prevalence and transmission of pathogens, e.g. *Brucella* and morbillivirus, issues remain with the quantitative interpretation of strandings, e.g. what can be learned from the total number of strandings, their geographical distribution, the proportions of different causes of death and size or age composition of the sample. Thus, an increase in the number of strandings in an area could signal an increase in mortality rate, a shift in distribution or increased abundance.

Standard protocols have been developed for carrying our necropsies and methods to diagnose different causes of death have improved over time. However, while general indicators exist for diagnosis of bycatch during pathological investigations, there is almost always a component of interpretation. Ideally, suspected bycatches among strandings need to be determined by experienced pathologists since mis- or overinterpretation can easily occur. Misinterpretation can originate from not recognizing signs of bycatch or by assuming that bycaught individuals are always healthy and therefore excluding the possibility that evidently sick animals had died due to bycatch. The procedures followed to obtain carcasses, diagnose causes of mortality and to assign a degree certainty to the diagnosis need to be detailed and, ideally, better harmonised. It is important that necropsy includes a full health assessment and not only identification of cause of death (Bernaldo de Quiros *et al.*, 2018; see also SMASS reports).

The identification of "suspected bycatches" among strandings evidently provides a minimum estimate of the number of animals dying due to bycatch. Furthermore, suspected bycatches among strandings also have a sentinel role as a valuable indicator of bycatch "hot spots", i.e. areas where fisheries interactions may be problematic (e.g. Baltic Sea, Bay of Biscay) and may help to initiate further monitoring and mitigation actions in those areas. For example, two unusual multiple stranding events totalling ca. 700–800 common dolphins were recorded in February–March 2017 along the French Atlantic coast, and 65% of the necropsied animals were diagnosed as bycaught (Peltier *et al.*, 2017). Another similar mass stranding has taken place during the WGMME meeting in February 2019. In addition to providing data and samples for a range of other studies (e.g. on life history, health, etc.), in relation to fishery bycatch, strandings can:

- provide information on which segments of the population may be most affected (e.g. young males, female + calf pairs, etc.);
- provide information on diet to assess prey targeted by bycaught animals;
- provide indications of the type of fishing gear involved;
- permit assessment of the welfare implications of chronic large animal entanglements in active or discarded fishing gear.

Notwithstanding the above points, strandings data have generally been considered to be unsuitable for calculating bycatch rates (e.g. ICES, 2011) due to the fact that the carcasses reaching the coast are unlikely to be representative of mortality at sea. Known sources of potential bias include factors that determine where, when and how many animals die, then whether (and if yes, with what delay) a dead animal (i) reaches the coast, (ii) is discovered, (iii) is necropsied.

The occurrence of mortality (amount, time, location) relates to both the abundance and distribution of animals in the sea (in turn related, for example to fish distribution and abundance), and the importance and distribution of different causes of mortality (e.g. fishing activity and (in the case of harbour porpoises) due to bottlenose dolphins).

Arrival of carcasses on the shore will depend on proximity to the coast at the time of death, the size and condition of the carcass (affecting buoyancy), wind conditions and currents, as well as coastal geomorphology. Related considerations include the residence time of a carcass on the shore: will it be carried away again at high tide and how rapidly will scavenging and decomposition render it unsuitable for necropsy? Research on carcass drift (e.g. Peltier *et al.,* 2012) is ongoing although evidently it does not yet provide a complete answer. An IWC Special Working Group reviewed the drift modelling work that has been undertaken to date, further reviewed by a subgroup of the IWC HIM Sub-Committee. Recommendations included addressing uncertainties in the analysis arising from parameters not yet directly quantified or with estimates based on limited and/or potentially biased samples. The group also highlighted uncertainties in the estimation of immersion level, the probability of being buoyant, the probability of stranding and the time of death (IWC, 2018). IWC (2018) further noted "*the importance of observer programmes, including electronic monitoring, and the limitations of stranding information for determining the type of fishing gear implicated in a bycatch event, or in determining reliable bycatch estimates*". However, the intense scrutiny given to inferences about bycatch from strandings has arguably not been matched in relation to other data sources and the fact that almost no robust estimates of cetacean bycatch rate exist for European fisheries 15 years after the implementation of 812/2004 (and this is without considering biases due to self-selection of participating vessels and modification of fisher behaviour when observers are on-board) underlines the importance of developing alternative methods to derive reliable estimates, for example from strandings data.

The probability of a stranded animal being discovered will depend on factors such as the nature of the strandings monitoring network, local human population density, coastal geomorphology, weather, condition of the carcass and awareness of/interest in the strandings monitoring programme. Stranding networks are in place in many European countries. However, search effort varies around the coast, geographically and seasonally, reflecting coastal morphology, human population density and local climate. In addition, public awareness of carcasses and stranding networks have grown over time. Decisions on whether an animal is necropsied may depend on its accessibility, budgetary constraints and specific research/monitoring priorities, which may result in unintended biases in the sample composition. Some of these issues require further study but others may be readily addressed by consulting widely available information and by more detailed/transparent record keeping. Improvements in citizen science reporting of strandings (while discouraging handling of dead animals due to safety issues, e.g. disease transmission) and the increase in digital images submitted from strandings have allowed for suspected bycatch cases to be better identified (data from Scotland).

Given the considerable attention paid to developing and harmonising necropsy protocols (e.g. a workshop is planned for the forthcoming World Marine Mammal Conference), reporting procedures (e.g. within ASCOBANS) and carcass drift models, the weakest links in the chain are probably understanding why and when carcasses are discovered and reported, as well as explicitly accounting for the decision process that leads to a necropsy being undertaken. In Scotland, SMASS will launch an app in May 2019, which will log time spent by members of the public searching for stranded animals along the coast, in order to build up a picture of effort distribution (www.beachtrack.org).

If the above-mentioned biases can be adequately accounted for, further use can be made of stranded animals if age data are available and the proportion of bycaught animals is known. Life table methods permit estimation of overall annual mortality rate, age and sex distribution, death age and data of pathological investigations allow this to be partitioned between causes of death (e.g. Read, 2013). Model-based approaches can be used to adjust the mortality estimate for underrepresented age classes (Saavedra, 2018), while drift modelling could help to correct the data for biases in the origin of the carcasses as well as helping to determine the size of the "catchment area" (which in most cases would not include the whole range of the population; e.g. Peltier and Ridoux, 2015). A low average age of death among stranded animals (compared with a reference value, e.g. from another population) may indicate high anthropogenic induced mortality (see Kesselring *et al.,* 2017).

To interpret results on mortality due to fishery bycatch, knowledge of the distribution of fishing activity is evidently important, but the wider context is also relevant, i.e. ideally we need to know more about the distribution of all natural and anthropogenic threats leading to cetacean mortality, including factors which might increase the risk of animals being bycaught (e.g. age class, prey depletion, poor health, ototoxic chemicals, acoustic interference or auditory impairment).

If bycatch information from strandings is to be pooled together across countries to obtain bycatch estimates for species/populations/MUs, it is crucial that the information is collected and reported in a standardised way. This is particularly important because the identification of health status and the determination of the cause of death is directly dependant on the quality of investigations and the experience of the investigating pathologist with the target species. Thus, ASCOBANS Member States report the identification of bycatches with different degrees of certainty, e.g. "probable bycatch", "plausible bycatch", etc. It is also important to separate bycatch mortality diagnoses from strandings and carcasses handed in by fishermen, which also need to be verified by a pathologist (for exclusion of carcasses drifting into the nets). In many countries, regulations prevent the intentional landing of bycaught cetaceans, so discarded and subsequently stranded animals represent the only legal source of necropsy data for bycatch cases.

The ASCOBANS North Sea Group has asked MS to compile bycatch information obtained from the examination of stranded porpoises in a common format (See Table 1). WGMME notes that this will be a useful way of ensuring consistency in the reporting on the available information on bycatch that can be derived from strandings data. To illustrate this point, in the following section, brief reports from some ICES member states are given.

Table 1. Overview of harbour porpoise strandings, necropsies and bycatch determination for the North Sea (input provided by Belgium, The Netherlands, Germany, Sweden and the United Kingdom). This table follows the format of the ASCOBANS North Sea Group request.

Country	Year	Number of recorded strandings*	Number of porpoises necropsied**	Number of necropsied porpoises with			% of strandings necropsied	% Bycatch in	
				known cause of death	unknown cause of death	cause of death by-catch		all animals necropsied	animals necropsied with known COD
FR	2013	313	1	1^	0	0	0.3	na	na
FR	2014	181	10	3^	7	3	5.5	na	na
FR	2015	131	6	5^	1	3	4.6	na	na
FR	2016	262	2	2^	0	1	0.8	na	na
FR	2017	168	1	1^	0	1	0.6	na	na
BE	2016	137	116~	33^	83	21	84.7	18.1	63.6
BE	2017	94	85~	25^	60	9	90.4	10.6	36.0
NL	2014	582	57	24	33	2***	9.8	3.5	8.3
NL	2015	309	32	28	4	1***	10.4	3.1	3.6
NL	2016	661	68	54	14	2***	10.3	2.9	3.7
DE	2015	109	109	-	-	3****	100****	2.8	2.8
DE	2016	126	126	-	-	2****	100****	1.6	1.6
DE	2017	91	91	-	-	5****	100****	5.5	5.5
SE	2016	19	4	3	1	1	21.1	na	na
SE	2017	19	20	6	1	1	30.0	na	na
UK	2016	248	39	39	0	1	15.7	2.6	2.6
UK	2017	185	33	33	0	1	17.8	3.0	3.0

*** Some databases include: live strandings that do not survive, also "partial" finds of porpoises and/or bones.**

**** Animals that were bycaught and brought in by fishermen were not included here.**

***** Cause of death (COD) codes included were: hpr (high probability of bycatch) and pr (probable bycatch). Animals considered to be possible bycatch are not considered.**

****** All strandings undergo a post-mortem examination but not necessarily a full necropsy.**

^ The database also includes animals with known cause of death that were not necropsied. These animals are not included here.

^^ Numbers not final.

~This includes animals for which the cause of death was determined without a necropsy.

Na Not applicable as sample sizes too small to be representative.

Additional remarks:

- The percentage of animals stranded that are necropsied varies greatly between countries. The highest percentage is for Germany (where all strandings undergo post-mortem examination but may not receive a full histological investigation including ears and brain) and Sweden (where relatively few strandings are recorded). For the remaining countries, between 10 and 20% of stranded animals are necropsied.
- Bycatch rates are similar for the UK and the NL. However, they are much higher for Belgium. These differences need explaining. The sample sizes for Sweden are too small to draw firm conclusions.
- The difference in numbers of recorded porpoise strandings between the UK and the Netherlands is striking, with many more in the NL despite its much shorter length of coastline.

Sweden: In Sweden, opportunistic observations of live or dead harbour porpoises can be reported to the Swedish Museum of Natural History (SMNH, financed by the Swedish Agency for Marine and Water Management) and to the Swedish Species Information Centre (SSIC, financed by the Swedish Environmental Protection Agency). The SSIC does not distinguish between live and dead animals in their database, but during 2016–2018, 38 harbour porpoise records included the word “dead”. During the same time period, a total of 226 harbour porpoises were reported as dead to the SMNH. Of these, two were encountered as live strandings, four were handed in as bycaught animals, and the remaining 220 were found as dead strandings. A total of 38 animals reported as dead strandings to the SMNH were collected and necropsied for health assessment, determination of cause of death, and sampling. Primarily only fresh animals were collected from the Skagerrak, Kattegat and Sound, while all animals that can be collected from the Baltic Sea are of interest to be collected. In a few cases, gross measurements and samples were taken directly on the beach. An overview of the animals reported to, collected and/or sampled by the SMNH is given in Table 2.

Given all caveats by assessing bycatch information from stranded animals, the results shall be interpreted with caution. However, it is worth noting that most of the dead strandings were found within the summer management area of the Belt Sea population (definitions by Sveegaard *et al.*, 2015), and the cause of death was determined as bycatch for more than half of the animals for which the cause of death could be determined.

Table 2. Sweden: Dead harbour porpoises reported to, collected and necropsied by the Swedish Museum of Natural History in 2016–2018.

Pop[1]	Live strandings[2]	Given as bycatch[3]	Dead strandings		Necropsied dead strandings with cause of death			% bycatch of known cause of death
			Total reported	Total necropsied	Known	Bycatch	Unknown	
NS	0	4	55	12	8	2	4	25%
BeS	2	0	156	26	15	11	11	73%
BaS	0	0	9	0	0	0	0	-
Total	*2*	*4*	*220*	*38*	*23*	*13*	*15*	*57%*

[1] Population management borders according to Sveegaard *et al.* (2015). NS = North Sea, Bes = Belt Sea, BaS = Baltic Sea.

[2] The live strandings died during rescue attempts and were collected for necropsy.

[3] The animals given as bycatch were collected for necropsy.

Poland: In 2018, strandings of harbour porpoises were continuously collected by Professor Krzysztof Skóra Hel Marine Station, University of Gdańsk. Cooperation with WWF Poland within the external project the "Blue Patrol" helped to collect 14 carcasses of harbour porpoise. The state of their decomposition was too advanced to let the post-mortem analysis be performed. The marks of human intervention (fins cut off; dorsal part cut) were observed in three of them. In addition, one porpoise bycaught in a gillnet was reported by fishermen operating in the central part of Polish coast.

Ireland: There is a marine mammal necropsy project currently running and augmented sampling scheme under the data collection framework (both are funded by EMFF through the Marine Institute), with additional sampling effort allocated to those fisheries that may pose a risk of cetacean and seal bycatch (and other PET species). Part of this work is to assess bycatch in stranded carcasses, work undertaken by the Regional Veterinary Laboratory, IWDG and GMIT, and also the Institute of Zoology, London (IoZ). The necropsy project was started in June 2017 and follows all UK Cetacean Strandings Investigation Programme (CSIP) necropsy protocols. The IoZ reviews all case history reporting as part of the project.

3.3.3 Remote monitoring (REM)

In the Electronic Recording and reporting System (ERS), fishers can fill in data on bycatches. REM systems have been considered to be one of the best solutions for controlling and monitoring fisheries operations. By installing CCTV cameras and sensors on the winches/drum and pump mechanisms, footage of the fishing operation is obtained and stored and can be later visualised by inspectors onshore. Generally, not all footage is reviewed, only a percentage of the fishing operations which is considered sufficient. The advantage of these systems in addition to the monitoring of the fishing activity is the deterrent effect of having the system in place. However, among its disadvantages, there is an issue of cost, and who should bear it and because of this and issues related with privacy, fishermen are fiercely opposed to its implementation that they perceive as a criminalisation of their activity.

Public and political pressure is growing to implement REM systems on vessels in fisheries where discarding is considered to be an issue. A key perceived benefit of REM systems when compared with deployment of observers on board is cost-effectiveness, because, although installation of REM systems implies significant costs, in the long term, it is estimated that maintenance costs (and the costs of viewing and analysing of the video footage), will be lower than the cost of paying for observers. If REM systems were in place, monitoring of bycatch could also take place as several examples have shown (e.g. Kindt-Larsen *et al.*, 2016; Scheidat *et al.*, 2018).

For seals, it is possible to assess the occurrence of non-fatal entanglement in fishing gear using REM systems deployed at haul-out sites, for example using unmanned aerial vehicles (Martins *et al.*, 2019).

3.3.4 Integrated approaches

Rubsch and Kock (2004) provided an estimate of porpoise bycatch along the German Baltic Sea coast. They included data from stranded bycaught animals and bycatch reported directly by fishermen, as well as data from interviews with the local fishing community from the years 1996 to 2002. They concluded that an estimated annual mean of 57 animals are caught in the German western Baltic and 25 animals in the German Baltic Proper.

3.4 (C iii) High risk areas, knowledge gaps for marine mammal bycatch

3.4.1 Background and rationale

In 2013, in response to a request from the EU, ICES advised the use of a Bycatch Risk Approach (BRA) to identify areas and fisheries which pose the greatest likely conservation threat to cetacean species due to bycatch (Figure 3; see ICES Advice, 2013)

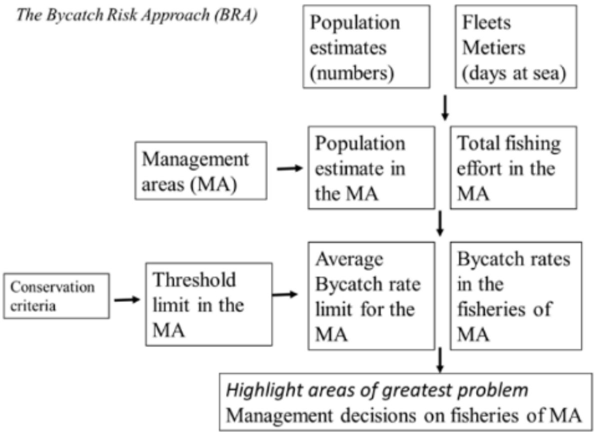

Figure 3. The Bycatch Risk Approach (BRA).

In 2014, WGBYC (ICES, 2014a) discussed ways to make progress with assessing protected species bycatch, drawing on results of a workshop in 2010 (ICES, WKRev812 2010) highlighting the value of compiling existing data on fishing effort and combining this information with available bycatch rate estimates to provide an indication of whether or not current levels of fishing effort might pose a conservation threat, specifically by:

i. Comparing a range of 'possible' or likely overall bycatch rates and effort data with some conservation reference level (e.g. 1.7% of best abundance estimate for the region (after ASCOBANS); or
ii. The BRA: given a species abundance estimate, a bycatch reference limit, and an estimate of total fishing effort, one can ask what overall bycatch rate would be needed to exceed the bycatch reference limit and then decide whether or not this is feasible.

WGBYC (ICES, 2015b) summarised estimates of porpoise bycatch rate, by region, using data from 2006–2013.

3.5 (C iv) Thresholds for marine mammal bycatch

3.5.1 Background

Through Regulation 812/2004 (cetaceans) and both the EU Habitats Directive and the Marine Strategy Framework Directive (MSFD) (seals and cetaceans), there is a requirement to assess and mitigate marine mammal bycatch mortality and ensure that it does not negatively impact on Favourable Conservation Status (FCS) or Good Environmental Status (GES).

Three general approaches for setting "safe bycatch limits" (threshold values) have been discussed in several fora (see, e.g. ICES Advice 2013; ICES, 2013a; ICES, 2013b; 2014b; ICES, 2013c; ASCOBANS, 2015a, b); namely fixed percentages of abundance, potential biological removal and the Catch Limit Algorithm.

In 2009, ICES advised the European Commission that "a Catch Limit Algorithm approach is the most appropriate method to set limits on the bycatch of harbour porpoises or common dolphins" while highlighting that in order to use this (or any other) approach, specific conservation objectives must first be specified. The latter point was reiterated in 2010: "ICES advised in 2009 of the need for explicit conservation and management objectives for managing interactions between fisheries and marine mammal populations. This advice has not been acted upon. Lacking these objectives, ICES is unable to properly consider the impacts of these interactions in its management advice" (see ICES, WGBYC 2013) for further discussion.

3.5.2 Percentage of abundance

Perhaps the best known proposed bycatch threshold, i.e. a limit above which bycatch levels are expected to cause population decline or at least prevent population growth and are therefore considered to be "unacceptable", was proposed by ASCOBANS (2000) for harbour porpoise, which defined "unacceptable interactions" as being, "in the short term, a total anthropogenic removal above 1.7% of the best available estimate of abundance". This is designed to meet the ASCOBANS interim objective to restore populations to, or maintain them at, 80% or more of carrying capacity over an infinite time horizon, assuming no uncertainty in any model parameter (IWC-ASCOBANS, 2000). This value has been used by a number of MS to assess whether the populations of marine mammals, not only harbour porpoise, could be considered to be in GES following the MSFD (2008/56/EC) requirements. For example, France has done so for both harbour porpoise and common dolphin (Spitz *et al.*, 2018).

This type of threshold, i.e. expressing the maximum annual removal as a proportion of the population abundance, is easily understood, although identifying an appropriate threshold value depends on knowledge of population dynamics and the ability to estimate the carrying capacity, which is a very difficult parameter to estimate. In practice, its use implies the need for (ideally) accurate and precise estimates of the number of animals bycaught and population size. Given that there is typically considerable uncertainty around both estimates, one solution is to set lower (precautionary) bycatch mortality limits. Thus, the Bergen Declaration set a "precautionary" maximum bycatch rate for harbour porpoises at 1%.

This approach could be refined by incorporating knowledge of age structure and age-specific mortality and fecundity. However, such data are often not available for the populations under study and indeed, under the MSFD, Member States are often considering Management Units (MUs) which do not correspond to natural populations.

Saavedra (2018) estimated bycatch limit reference points for common and bottlenose dolphin populations inhabiting the continental shelf Atlantic waters of the Iberian Peninsula using a minimum realistic ecosystem model implemented with GADGET. The author reported bycatch limit reference points of 1.4% of the best available abundance estimate for both species, noting that the calculated confidence limits indicate that the bycatch limits should be reduced to 0.7% for common dolphins and to 0.9% for bottlenose dolphins to be precautionary. The author discussed that the lower rates obtained for common and bottlenose dolphins when compared with harbour porpoise could be related to the fact that harbour porpoises could sustain higher levels of bycatch due to their earlier maturation and higher pregnancy rate.

3.5.3 Potential Biological Removal

The Potential Biological Removal (PBR) approach defines the maximum number of animals, in addition to natural mortality, that may be removed from a stock while allowing it to reach or maintain its optimum sustainable population (at or above the level that will result in maximum productivity). This approach is applied in the USA under the Marine Mammal Protection Act.

The PBR level is the product of: (1) the minimum population estimate; (2) one-half the maximum theoretical or estimated net productivity rate; and (3) a recovery factor of between 0.1 and 1.0. The value of the recovery factor is determined by information on the status of the stock. If the stock is declining and/or endangered, it is set to 0.1. If there is no information on status, it is set to 0.5. Values between 0.5 and 1.0 are set for other stocks depending on the information available (ICES, WGMME 2014).

In December 2018, NAMMCO and the Norwegian Institute for Marine Research carried out an assessment of the harbour porpoise management units in the Northeast Atlantic (North Atlantic Marine Mammal Commission and the Norwegian Institute for Marine Research 2019). For most assessment units, a deterministic density-dependent sex- and age-aggregated population dynamics production model (Zerbini *et al.*, 2011) was used. However, for assessment units that did not have sufficient data to include in the population dynamic model, status of the assessment unit was evaluated by comparing bycatch levels to the PBR level (Wade *et al.*, 1998).

3.5.4 Catch Limit Algorithm (CLA)

This approach is used within the IWC´s Revised Management Procedure (RMP). Put very simply, the CLA fits a population dynamics model to a time-series of abundance estimates and removals data whereas PBR uses a single value of minimum abundance. In 2009, ICES advised the European Commission that "a Catch Limit Algorithm approach is the most appropriate method to set limits on the bycatch of harbour porpoises or common dolphins" (ICES, 2013c). ICES, WKBYC (2013) noted that the existing procedures to establish limits and reference points (CLA, PBR, and the fixed 1.7%) have been reviewed several times in the past decade. WKBYC (2013) further noted the recommendations made by the WGMME (ICES, 2013b), particularly that the CLA approach is recognised as the most appropriate method to set limits on the bycatch of cetaceans. These reports also highlighted that any approach will need specific conservation objectives that should be adopted at European level, and noted that improved data on bycatch, fishing effort and the biology of the species would improve the procedure (ICES, 2013c). Hammond *et al.* (2018) developed a Removals Limit Algorithm (RLA) for harbour porpoises in the North Sea that, similarly to the CLA, allows the setting of limits to the anthropogenic mortality (in this case, bycatch) that a cetacean population can withstand, if specified conservation objectives need to be met.

3.6 Other considerations

Unbiased population abundance estimates are essential to permit determination of bycatch thresholds, but large-scale surveys are typically carried out during a single calendar month. Rogan *et al.* (2017) demonstrated marked seasonal differences in cetacean abundance around Ireland, with larger numbers of animals recorded in winter in the ObSERVE project surveys. This was also the case of common dolphins in the Bay of Biscay as shown by the results of the SAMM surveys (e.g. Laran *et al.*, 2017).

These seasonal differences in abundance estimates raise the question as to which abundance estimates should be used to assess the impact of bycatch.

For any source of bycatch information to be useful for conservation and management purposes, once biases and uncertainties have been taken into account, it should be possible to aggregate it by Management Unit / population – implying that national data will in some cases need to be subdivided (e.g. for populations of coastal bottlenose dolphins) and in many cases, data will need to be aggregated across countries.

Evidently, for this to be possible, common (verifiable) procedures of data collection are needed and all data must be georeferenced.

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4 ToR D. Review the population-level effect of cumulative human impacts on marine mammals and further develop and/or update the threats matrix

4.1 Cumulative Impact Assessments (CIA)

ICES (2014) concluded that "there is little information on how multiple environmental pressures might collectively impact marine mammals. […]. The cumulative effects of these disparate industries [(shipping, fisheries, offshore oil/gas exploitation, marine renewable energies, etc.)] on the long-term conservation status of marine mammal populations are only just beginning to be explored."

The US National Academy of Sciences, Engineering, and Medicine recently published a comprehensive report on the state-of-the-art understanding of cumulative effect of pressures on marine mammals (NASEM, 2017). A brief summary of this report is provided below with some clarification of the terminology used.

There is an important distinction to make between a hazard, which is the potential for harm (e.g. mercury is hazardous), and risk, which relates to the actual potential for harm after mitigating actions have been taken (Sheer *et al.*, 2014; Spiegelhalter, 2017). Risk is often considered as a function of both the probability of an adverse outcome and the magnitude of the consequences (Spiegelhalter, 2017). In short, the potential to harm an individual (or the environment) is defined as a hazard, whereas risk also encompasses the probability of exposure and the magnitude and/or extent of damage (that is the impact of a harmful event on an individual, population or ecosystem, for example).

Another important distinction in cumulative assessments relates to what is actually being accumulated: risks (i.e. exposure to hazards) and/or effects of pressures (i.e. their impacts). For marine mammals and other apex predators, cumulative risk assessments have been carried out at rather large even global scales (e.g. Maxwell *et al.*, 2013; Avila *et al.*, 2018). In these assessments, cumulative risk derives from the combination of noise and other anthropogenic pressures, such as chemical pollution, marine debris, introduced pathogens, fishing, as well as natural pressures such as increased presence of predators, pathogens, parasites, or reduced availability of prey due to natural or human-induced ecological interactions (NASEM, 2017, p 1). However, the quantitative prediction of the cumulative effect of pressures is currently not possible (NASEM, 2017). NASEM (2017) thus defines "aggregate exposure as the combined exposure to one stressor from multiple sources or pathways and cumulative effect as the combined effect of exposures to multiple stressors integrated over a defined relevant period: a day, a season, a year, or a lifetime" (pp. 11–12). The ultimate goal of cumulative assessments is to have predictive values that managers can use to decide when (and where) those combined effects are particularly important.

NASEM (2017) outlined a conceptual model of Population Consequences of Multiple Stressors (PCoMS, Figure 4), which generalizes the Population Consequences of Disturbance (PCoD) model to multiple pressures. The PCoD model is a conceptual framework that formalises the links from exposure to a pressure (e.g. noise; Costa *et al.*, 2016; Nabe-Nielsen *et al.*, 2018) to individual-level changes in behaviour and physiology, which in turn can effect individual health and vital rates, and ultimately population dynamics (Pirotta *et al.*, 2018). Modelling health enables to scale responses from the individual to the population level. An individual's health encompasses many aspects of its physiology, which can be difficult to measure in practice. To date, most PCoD applications have used an individual's energy stores (that is, its body condition) as the measure of health (Pirotta *et al.*, 2018), or have developed a bioenergetic model (Berajano *et al.*, 2017; McHuron *et al.*, 2018; Villegas-Amtmann *et al.*, 2015). When relevant data on physiology or health are lacking or unavailable, a pragmatic approach is to link directly behavioural responses to vital rates, with the provision that the resulting interim PCoD model (iPCoD, Pirotta *et al.*, 2018) will be updated once adequate data are collected. Expert elicitation may be used in iPCoD to inform parameters (King *et al.*, 2015).

Assessing the effects of a single pressure (e.g. noise) on behaviour, physiology and health of individuals is clearly challenging (see Pirotta *et al.*, 2018 for a recent review and some of its application on marine mammal populations). The PCoMS model is intended to take one step further, but the scientific challenges are considerable, especially with respect to sublethal effects, and (synergistic or antagonistic) interactions between multiple pressures. NASEM (2017, p. 40) states, "linking chemical stressors to decreases in vital rates through observational assessments is inherently challenging due to the chronic nature of many exposures or effects, the complexity involved in controlling for confounding or interacting variables, and the difficulty of observing mortality or reproductive endpoints in long-lived marine mammal species, particularly cetaceans. These challenges extend to other stressors that induce sublethal effects. Regardless of the stressor, few studies have explicitly defined quantitative relationships between varying doses and associated mortality, reproductive, or physiological effects for marine mammals."

Challenges remain in the proper calibration and estimation of exposure-response models with several pressures (NASEM, 2017, p. 51). A technical difficulty highlighted by NASEM (2017) is the statistical assumption of additivity when modelling interactions. Greenland (2009, p. 14) stressed that "concepts of biologic interaction do not in general correspond to the concept of statistical interaction, because the latter is only the need for a product term in a statistical model". The assumption of additive effects of multiple stressors is bold and may lead to an underestimation or overestimation of their cumulative impact. Given that most marine mammal populations are actually exposed to multiple stressors, a truly precautionary approach to their conservation should factor in the potential for synergistic (or antagonistic) cumulative effects of multiple stressors.

The common currency upon which cumulative effects are to be measured in PCoMS boils down to individual health and vital rates, such as survival or fecundity. There are challenges in collecting the necessary data to assess these vital rates and the impact of multiple pressures upon these. An additional challenge is the scaling up needed to move from individuals to population at the relevant spatio-temporal scales to provide accurate and useful predictions of cumulative effect. Nevertheless, PCoMS provides an idealized common conceptual framework to strive for quantitative assessment of cumulative effects of pressures. There is an urgent need to fund and foster interdisciplinary efforts to develop methods for estimating dose-response functions for high priority, ecologically relevant pressures, for studying how these pressures interact to generate impacts, and for developing theory and methods to predict how different pressures causing different kinds of impacts will interact to affect vital rates.

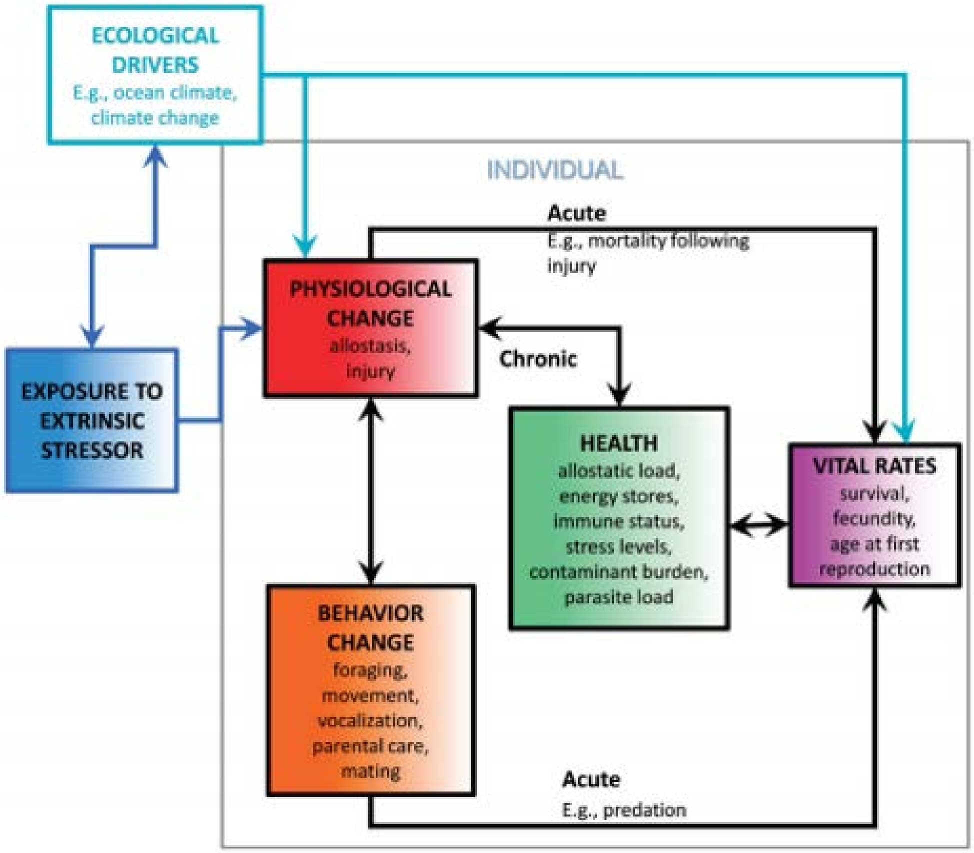

Figure 4. Population Consequences of Multiple Stressors (PCoMS) conceptual model (reproduced from NASEM 2017).

4.1.1 Bow tie diagrams for Cumulative Impact Assessment (CIA)

The broad application of bow tie diagrams for assessing cumulative effects on indicators was presented in the Report of the OSPAR ICG Eco-C workshop (OSPAR, 2018). Subsequently, the potential for applying this approach to marine mammals was discussed at the OSPAR-COBAM OMMEG meeting (February 2019), using contaminants as a working example (based on Figure 8 in the aforementioned report). Where OMMEG saw value in the approach, it was noted that, for marine mammals, the challenge is often the lack of ability to assess what is a direct or indirect impact, that these are unlikely to be independent of each other, and that the magnitude of the impact will vary between these potential pathways. This would ultimately lead to a complex, and somewhat difficult to interpret bow tie diagram, unless certain aspects of the assessment are packaged more broadly, as is suggested in the worked example in the aforementioned report. As such, bow tie diagrams may assist in basic visualisation of a cumulative effects assessment when qualitatively considering multiple assessments of impacts in isolation. However, these do not provide a method for quantitatively assessing cumulative effects of the multiple pressures/stressors that marine mammals are exposed to (the broad scope of which is outlined in the ICES threat matrices (see below)).

4.2 Threats matrix for marine mammals

In 2015, ICES WGMME was asked to compile a matrix of threats to the predominant marine mammal species in each of the MSFD regional seas (ICES, 2015). In accordance with ToR D, this has been developed and updated. The Baltic Sea has been split into two regions (Baltic Sea Proper and the Belt Seas & Kattegat), since the populations of harbour porpoise are considered to represent separate management units (Evans and Teilmann, 2009; Sveegaard *et al.*, 2015; ICES, 2018a), and the levels of threat from some human activities differ. Other regions identified here include the Greater North Sea (which includes the Skagerrak and the eastern half of the English Channel), the Celtic Seas (including the West of Scotland, Irish Sea, western half of the English Channel, and the seas west and south of Ireland), Bay of Biscay and Iberian Peninsula, and Macaronesia (Azores, Madeira, Canaries and Cape Verdes).

Following ICES, WGMME (2015), threat levels are classified as high, medium or low, adopting a traffic light system for each species-region combination, using the following criteria:

- High (red) = evidence or strong likelihood of negative population effects, mediated through effects on individual mortality, health and/or reproduction;
- Medium (yellow) = evidence or strong likelihood of impact at individual level on survival, health or reproduction but effect at population level is not clear;
- Low (green) = possible negative impact on individuals but evidence is weak and/or occurrences are infrequent.

The category “other” (no colour) is defined for cases where there was little or no information on the impact of these pressures on marine mammals or the threat is absent or irrelevant to a particular region-species combination. In both cases, it is indicated in the corresponding cell in the table.

Table 3. Descriptions of pressures by category, adapted from OSPAR ICG-COBAM (2012) and ICES (2015).

Pressure	Pressure description
Contaminants	Increase in the level of contaminants (transition elements, hydrocarbons and polycyclic aromatic hydrocarbons (PAHs), synthetic compounds, etc.), particularly those legacy compounds that persist in the environment and bioaccumulate through the food chain
Microplastics	Small (less than 5 mm diameter) plastic fragments show very low degradation rates and may contain chemical contaminants such as polybrominated diphenyl ethers (PBDEs), phthalates, flame retardants, etc., as well as act as carriers for the transfer of Persistent Organic Pollutants to organisms
Nutrient enrichment	Increased levels of nitrogen, phosphorus, silicon (and iron) in the marine environment
Habitat loss	Permanent loss of marine habitats, associated with land claim, new coastal defences, harbour developments, etc.
Habitat degradation	Overall damage to marine habitats and their ecosystems, due to seabed damage, fisheries, coastal/marine construction
Litter (including plastics and discarded fishing gear)	Marine litter includes: plastics, metals, timber, rope, fishing gear, etc. which may cause suffocation or entanglement
Sonar	Active sonar operates at frequencies from 0.1->150 kHz. Here we analyse effects of medium frequency (1–10 KHz) sonar used in military activities
Seismic surveys	Seismic surveys use airguns to generate loud low frequency sounds (largely <100 Hz) to explore the structure underneath the seabed; seismic vessels can also emit high frequency sounds
Pile-driving	Pile-driving is considered one of the loudest sources of low frequency (<100 Hz) pulsed sounds
Explosions	Explosions used in marine construction & military ordnance removal are very loud and of low frequency (<50 Hz)
Shipping	Noise from ships has a low frequency range, i.e. less than 1 kHz, although small pleasure vessels generate higher frequency sounds due to propeller cavitation. There has been a large increase in ambient noise in recent years, particularly in the northern hemisphere.
Barrier to species movement (offshore windfarm, wave or tidal device arrays)	The physical obstruction of species' movements, including local movements and regional/global migrations
Death or injury by collision (with ships)	Injury or mortality from collisions of biota with moving structures
Death or injury by collision (with tidal devices)	Injury or mortality from collisions of biota with static structures
Introduction of microbial pathogens	Untreated or insufficiently treated effluent discharges and run-off from terrestrial sources and vessels
Removal of target and non-target species (prey depletion)	The commercial exploitation of fish & shellfish stocks, including smaller scale harvesting, angling and scientific sampling. Ecological consequences include the sustainability of stocks, impacting energy flows through foodwebs and the size and age composition within fish stocks
Removal of non-target species (bycatch)	Bycatch associated with all fishing activities. It addresses the direct removal of individuals associated with fishing/harvesting. Ecological consequences include foodweb dependencies, population dynamics of fish, marine mammals, turtles and seabirds

Pressure	Pressure description
Disturbance (e.g. wildlife watching)	Disturbance of biota by anthropogenic activities, e.g. increased vessel movements, increased personnel movements, increased tourism, increased vehicular movements on shore, etc.
Deliberate killing + hunting	Marine mammals are killed in some countries for food or when perceived in conflict with human activities such as fishing and aquaculture

Marine mammals face a variety of human pressures. These are described in Table 3, with more detailed information on each in Section 4.2.1, where summaries are provided on the evidence of impacts on particular species and regional variation is assessed. Tables 4–9 present assessments of the threat levels for all the main pressures in each of the six regions, according to species. Only those species regularly occurring in a particular region are assessed. Assessments were undertaken by WGMME in consultation with regional experts (see acknowledgements and member directory), using the criteria described above. Justification for the assessments is detailed below, along with explanations for any changes since 2015. In most cases, changes have resulted from increased information rather than from a change in a particular human activity.

4.2.1 Human activities and their potential impacts upon marine mammals

4.2.1.1 Pollution and other chemical changes

Contaminants

A wide range of contaminants are known to have negative effects upon marine organisms (OSPAR, 2010; 2017; HELCOM, 2010; 2018). Several of these have been implicated as affecting marine mammals (e.g. Jepson *et al*., 2016; Fossi and Panti, 2018). However, PCBs in particular have been shown to negatively affect reproduction as well as resistance to disease (Reijnders, 1986; Jepson *et al*., 2005; 2016; ICES, 2010; Murphy *et al*., 2015; 2018; Lehnert *et al*., 2018; Robinson *et al*., 2019), and species with the highest levels in Europe appear to be killer whale, bottlenose dolphin, striped dolphin, and harbour porpoise (Jepson *et al*., 2016). PCB levels in common dolphins have also been shown to negatively affect reproduction (Murphy *et al*., 2018). Baleen whales generally have lower contaminant levels, presumably because they are feeding at lower trophic levels (O'Shea and Brownell, 1994; Hobbs *et al*., 2003). Highest levels of PCBs in the marine environment occur in the Baltic Sea (HELCOM, 2010; 2018), and Greater North Sea, followed by the Celtic Seas and Bay of Biscay (OSPAR, 2010; 2017). Along the southern Iberian Peninsula, the declining Strait of Gibraltar population of long-finned pilot whales has levels of PCBs much higher than in either the NW Mediterranean or NE Atlantic, and above the threshold considered harmful (Verborgh *et al*., 2016). Although not monitored to the same extent, levels of PCBs in bottlenose dolphins from the Canary Islands are at toxicological levels that give rise to concern, and those levels have been increasing since 2008 (García-Alvarez *et al*., 2014). Elevated levels of persistent organic pollutants have also been reported for common dolphin and Atlantic spotted dolphin in the Canaries and Azores (Méndez-Fernandez *et al*., 2017; 2018).

Baltic grey and ringed seals have carried high loads of organochlorine and heavy metal contaminants that have been implicated in uterine pathology (Bergman and Olsson, 1986), and in the 1970s, only 17% of females were thought to be fertile (Helle, 1980). Although levels in Baltic seals have declined since restrictions came into force on use and release of persistent organic pollutants, fertility rates of ringed seal females are only about 68% in the Bothnian Bay, below the rates found in some Arctic populations (HELCOM, 2015).

Nutrient enrichment

The southern North Sea and English Channel receive high levels of phosphates and nitrates from freshwater outflow from major river systems (OSPAR, 2010; 2017). Similarly, the Kattegat, the Danish Straits, the Gulf of Finland, the Gotland Basin as well as main parts of the Arkona Basin, the Bornholm Basin, and the Baltic Proper have been classified as problem areas (HELCOM, 2009), with only small signs of improvement in recent years (HELCOM, 2018). The consequences (which may be exacerbated by warming seas) can be massive algal blooms leading to die-offs of fish and invertebrates, which potentially could affect marine mammals. Although, to date, no obvious incidents have been reported, urine and faecal samples from harbour seals in the North Sea have tested positive for domoic acid, a toxin produced during harmful algal blooms (Hall and Fame, 2010; Jensen *et al.*, 2015; SCOS, 2018). Dinoflagellate brevetoxins have been linked to mass mortalities in bottlenose dolphins in southeastern United States (Fire *et al.*, 2008; Twiner *et al.*, 2011; Fire and Van Dolah, 2012) and in a sei whale mass mortality in southern Chile (Häussermann *et al.*, 2017).

Microplastics

There has been growing concern over potential detrimental effects of microplastics upon marine organisms including marine mammals, particularly filter-feeding species such as the fin whale (Fossi *et al.*, 2012; 2014; 2016; 2017). Microplastic fragments and nanoplastics show very low degradation rates and may contain chemical contaminants such as polybrominated diphenyl ethers (PBDEs), phthalates, flame retardants, etc, as well as act as carriers for the transfer of POPs to organisms. Recent studies have shown that, in seals, microplastics can be passed from mother to offspring through her milk (K. Bennett, *pers. comm.*). Microplastics may also potentially impact marine mammals through the effects of endocrine disrupting chemicals that they can carry. At present neither the levels nor the impacts of microplastics upon marine mammals are well understood.

4.2.1.2 Physical Loss

Habitat loss

Physical loss of habitat at sea occurs when structures such as wind turbines or oil and gas platforms are built. If those cover a large extent, they may have a negative impact on marine mammals by removing feeding habitat. On the other hand, they could serve as artificial reefs, attracting fish and invertebrates (Kjær *et al.*, 2006; Huddleston, 2010). Land reclamation schemes, coastal defences, industrial and building developments along coastlines may remove breeding or haul-out sites for seals, with the monk seal in the Mediterranean being particularly vulnerable (Karamanlidis *et al.*, 2016). However, no direct impacts of human activities on seal sites have been reported in recent years in the regions under review (OSPAR, 2010; 2017). A special case is the loss of sea ice breeding habitat for grey seals and ringed seals in the Baltic (in areas such as the Gulf of Riga and Bothnian Bay) as a result of climate warming (Meier *et al.*, 2004; Sundqvist *et al.*, 2012).

4.2.1.3 Physical damage

Habitat degradation

Marine habitats can be degraded by bottom trawling, sand and gravel extraction, and dredging activities damaging the seabed (OSPAR, 2010; 2017; HELCOM, 2018; Rijnsdorp *et al.*, 2018). Disturbances to the seabed from shipping may also occur, mainly in shallow areas. The effects are often local, concentrated to shipping lanes and in the vicinity of harbours (e.g. there is a large harbour construction within the Special Area of Conservation in SW Tenerife which may affect short-finned pilot whale and bottlenose dolphins that feed in the area, N. Aguilar de Soto, *pers. comm.*). Erosion of the seabed from abrasion can be substantial along heavy shipping lanes (Rytkönen *et al*., 2001). Eutrophication also causes habitat degradation through removal of oxygen for the water column. Historically this has been a major problem in deep waters of the central Baltic Sea (HELCOM, 2010; 2018). These may have negative effects indirectly upon marine mammals. There has been some small recovery since 2013 as a result of a series of inflow events improving oxygenation in the Arkona Basin, Bornholm Basin, and Eastern Gotland Basin, and the northern Baltic Proper (HELCOM, 2018).

4.2.1.4 Other Physical Pressures

Litter (including macroplastics and discarded fishing gear)

Marine litter has become a major issue of concern in the various regions under review (OSPAR, 2010; 2014; 2017; IWC, 2014; HELCOM, 2018). Impacts upon marine mammals appear to be greatest on deep diving cetaceans such as sperm whale and beaked whales that use suction feeding to capture cephalopods close to or on the seabed (Baulch and Perry, 2014; Lusher *et al*., 2015; Fossi *et al*., 2018), as well as other deep divers such as Risso's dolphin (Puig-Lozano *et al*., 2018). Although marine litter, including both macroplastics and microplastics, have been found in the stomachs of several necropsied cetacean species, there is little evidence so far that these caused their deaths (Fossi *et al*., 2012; 2014, 2018; Panti *et al*., 2019). However, in the Canary Islands, two out of twelve necropsied Gervais' beaked whales were determined to have died from ingestion of plastics (Fossi *et al*., 2018). Entanglement in discarded fishing gear, reported mainly from the UK, appears to particularly affect baleen whales such as the minke whale, (Northridge *et al*., 2010). Seals also are vulnerable to entanglement in discarded fishing gear (see, for example, Allen *et al*., 2012).

Underwater noise

Five main sources of underwater noise are reviewed here, with explosions added to those considered in the previous regional review of threats (ICES, 2015). Impulsive noise (sonar, seismic, pile driving, and explosions) and continuous noise (largely from shipping) sources are pressures being mapped for the Marine Strategy Framework Directive to fulfil assessment of Good Environmental Status for Descriptor 11 relating to Underwater Noise. A Marine Noise Registry has been established by ICES, and the updated threat matrices make use of information provided by EU Member States.

a) Sonar

Active sonar, operating with sound source levels of up to 235 dB re 1 μPa @ 1 m at frequencies mainly between 1 and 150 kHz (Hildebrand, 2009), is frequently used for fish-finding, oceanography, charting and in military activities (for example locating submarines). Since the mid-1990s, concerns have been expressed over the impacts that mid-frequency (1–10 kHz) sonar can have upon cetaceans, particularly beaked whales (Frantzis, 1998; Evans and Miller, 2004; Cox *et al.*, 2006; Tyack *et al*., 2011; Miller *et al*., 2015; Wensveen *et al.*, 2019; Bernaldo de Quirós *et al*., 2019). Effects include behavioural responses (at tens of kilometre range) and even ear damage and haemorrhaging in other tissues (Evans and England, 2000), as well as decompression symptoms leading to gas emboli (Fernández *et al*., 2004; 2005). Of the regions under review here, mass strandings of beaked whales associated with military sonar activities have occurred on a number of occasions in Macaronesia (Evans and Miller, 2004; Cox *et al*., 2006; Arbelo *et al*., 2013; Bernaldo de Quirós *et al*., 2019). Military sonar for submarine detection are deployed mainly in deep waters, which tend to be the main habitats of beaked whales, although dose-response studies suggest that other deep-water species such as killer whale and pilot whales may be vulnerable (Harris *et al*., 2015), and negative responses to sonar have been observed in minke whales (Kvadsheim *et al*., 2017) and captive harbour porpoise (Kastelein *et al*., 2015).

b) Seismic surveys

Seismic surveys for geophysical exploration for oil and gas started in Europe in the North Sea in the late 1950s, expanding to the European Atlantic and the Baltic in the 1970s (Evans *et al*., 2016). Seismic activities in the North Sea have reduced in the last ten years, but areas such as the Atlantic shelf edge (Celtic Seas) continue to be explored. Seismic airguns produce short duration broadband impulse sounds with source levels up to ca. 220–255 dB re 1 μPa peak-peak at 1 m, and most energy in the low frequency range below 300 Hz (Richardson *et al*., 1995). Several studies have reported negative behavioural responses to seismic sound at tens of kilometres range, mainly in baleen whales (whose communication signals and presumably hearing sensitivity are in the same low frequency range), with the risk of auditory injury at close ranges (Richardson *et al*., 1995; Nowacek *et al*., 2007; Southall *et al*., 2007). Small-scale negative reactions have also been observed from some toothed whales and dolphins (Stone *et al*., 2017). Seismic has been shown to affect foraging of sperm whales out to about 10 km range (Miller *et* al., 2009) and there are two cases of beaked whale strandings coinciding with seismic survey (Castellote and Llorens, 2015). So far, there is little evidence of effects on delphinids being anything but short term. Those species have hearing sensitivities concentrated in the high-frequency range (Southall *et al*., 2007). Although seismic airguns also produce some energy at high frequencies, transmission loss means that propagation is likely to be over shorter distances. So far, in porpoises, only short-term reactions to seismic airguns have been found (Thompson *et al*., 2013; Pirotta *et al*., 2014), although temporary hearing threshold shift has been found in a harbour porpoise after exposure to multiple airgun sounds (Kastelein *et al*., 2017).

c) Pile-driving

Pile-driving is typically used during the construction of offshore wind turbines (Kjær *et al*., 2006; Huddleston, 2010). The driven piles may give rise to peak-to-peak source levels in excess of 250 dB re 1 μPa @ 1 m at peak frequencies of 100–500 Hz, although >100 kHz can be produced (Tougaard *et al*., 2009a; Dähne *et al*., 2013). Several studies have reported negative effects of pile-driving on marine mammals, particularly harbour porpoise, at distances of at least 20 km, with animals leaving the area (e.g. Brandt *et al*., 2011; Tougaard *et al*., 2009a, b; Dähne *et al*., 2013; Mann and Teilmann, 2013), although most studies show that porpoises return after the construction phase (Brandt *et al*., 2018) and whether the area is being utilised by the species for feeding, breeding or simply to transit through is probably important. There is the potential for auditory injury (Kastelein *et al*., 2015). Most pile-driving activity occurs in shallow waters, in the Baltic Sea, Greater North Sea and Irish Sea (OSPAR, 2010, 2017; HELCOM, 2018). Only limited effects upon seals have been observed, with any avoidance movements being short term, although effects have still been reported at 20 km range (Tougaard *et* al., 2009b; Russell *et al*., 2016; Aarts *et al*., 2018), although harbour seals may be more vulnerable than grey seals due to their more restricted foraging ranges that bring them into greater contact with marine construction activities where the two overlap.

d) Explosions

Underwater explosions may be used in marine construction but also to remove unwanted structures on the seabed, including military ordnance. They are one of the strongest point sources of any man-made sound, starting with an initial shock pulse followed by a succession of oscillating bubble pulses (Richardson *et al.*, 1995; OSPAR, 2009). The source levels vary with the type and the amounts of explosives used, as well as the water depth at which the explosion occurs, and have ranged from 272 to 287 dB re 1 µPa zero to peak at 1m distance (for small explosives of 1–100 lb TNT). Frequencies are rather low with most energy between 6–21 Hz and lasting <1–10 ms (Richardson *et al.*, 1995; NRC, 2003). In coastal areas of the southern North Sea, disposal of unexploded military ordnance dumped during World War II has raised concerns over potential impacts upon porpoises (Koschinski, 2011; von Benda-Beckmann *et al.*, 2015).

e) Shipping

Shipping is considered the most widespread source of man-made sound in the ocean (OSPAR, 2009). Large vessels typically have sound source levels of 160–220 dB re 1 µPa @ 1 m over a bandwidth of 2–100 Hz (Richardson *et al.* 1995; NRC, 2003). The effects of shipping sound are largely unknown, but it clearly has the potential to mask communication, particularly amongst baleen whales that vocalise at frequencies that overlap with shipping sound and which may rely upon their vocalisations being heard over great distances (Clark *et al.*, 2009; Erbe *et al.*, 2016). Opportunistic study of stress hormones when shipping noise reduced after 9/11 also suggests that shipping noise may chronically elevate stress hormones in the endangered North Atlantic right whale (Rolland *et al.*, 2012). Even small cetaceans such as harbour porpoise have been shown to react negatively to shipping at distances ranging from hundreds of metres to a few kilometres (Dyndo *et al.*, 2015; Wisniewska *et al.*, 2018), whilst smaller vessels can cause avoidance reactions in porpoises, bottlenose dolphins and other coastal species (Evans, 1996; Würsig and Evans, 2001; Buckstaff, 2004; New *et al.*, 2013; Pirotta *et al.*, 2015). The busiest waterways in the regions under review are the southernmost part of the North Sea and English Channel, the southwestern Baltic and Danish Belt Seas, and across the outer part of the Bay of Biscay (OSPAR, 2010; 2017; Evans *et al.*, 2011; HELCOM, 2018).

There is little information at sea on behavioural responses by seals to shipping noise, but there is potential for masking of communication calls, disturbance and auditory damage where there is strong spatial overlap between seal occurrence and ship traffic (Jones *et al.*, 2017).

4.2.1.5 Barrier to species movement (offshore windfarm, wave, or tidal energy device arrays, oil or gas platforms)

Structures in the sea present potential barriers to movement for marine mammals. Impacts from these are not likely to be significant unless they occupy large areas, as may be the case for the larger windfarms that have been constructed in parts of the southern North Sea and western Baltic (Kjær *et al.*, 2006; Wilson *et al.*, 2007; Huddleston, 2010; Mann and Teilmann, 2013). In the case of seals confronted with a tidal turbine in the entrance to Strangford Loch (Northern Ireland), no barrier effect was found (Sparling *et al.*, 2018). In some cases, such structures may have a positive effect by serving as artificial reefs for invertebrates and fish, and they may lead to no-fishing zones.

4.2.1.6 Death or injury by collision

a) With ships

Collisions with vessels (particularly large ships travelling at speeds of 14–15 knots or more) have been reported for a number of cetacean species, but particularly fin whale, North Atlantic right whale, and sperm whale (Laist *et al.*, 2001; Pesante *et al.*, 2002; Vanderlaan and Taggart, 2007; IWC, 2008). Mapping of overlap between high shipping densities and high animal densities of the more vulnerable cetacean species listed above indicate the Bay of Biscay to be the area in NW Europe at greatest risk (Evans *et al*., 2011). In Macaronesia, sperm whales in particular are known to suffer ship strikes as well as fin whale, Bryde's whale, pilot whales and beaked whales (Ritter, 2007; Arbelo *et al*., 2013; Fais *et al*., 2016; Cunha *et al*., 2017; Dinis *et al*., 2017; Díaz-Delgado *et al*., 2018; N. Aguilar de Soto, *pers. comm.*). Small vessels travelling at speed may also cause physical injury (e.g. blunt trauma, propeller cuts), as revealed from photographs of coastal bottlenose dolphins (Feingold and Evans, 2014). Pinnipeds are assumed to be less vulnerable.

b) With tidal turbines

Concerns have been expressed over potential risk to seals and cetaceans of collisions with tidal turbines (Benjamins *et al*., 2015; Sparling *et al*., 2015; Onoufriou *et al*., 2019). Areas where turbines to exploit tidal energy have been deployed already include the Orkney Islands (Greater North Sea), Strangford Lough in Northern Ireland and off the coast of West Wales (Celtic Seas) with many areas proposed around the UK. Potential interactions have been anticipated particularly for harbour porpoise, which commonly utilise tidal stream environments, as well as bottlenose dolphins and seals (Benjamins *et al*., 2016; Waggitt *et al*., 2017). However, so far, despite several quantitative studies on this topic, no incidents have been reported, with tagged seals taking aversive action (e.g. Sparling *et al*., 2018; Joy *et al*., 2018).

4.2.1.7 Biological Pressures

Introduction of microbial pathogens

Microbial pathogens may be introduced into the marine environment from sewage discharges, transfer from fish or shellfish cultivated for mariculture, or even from humans or other terrestrial mammals (OSPAR, 2010; 2017; HELCOM, 2018). Potentially, these can all be the source of transmission of disease to marine mammals (Mazzariol *et al*., 2018). The most obvious examples are morbilliviruses that have resulted in seal epizootics in the North and Baltic Seas (Siebert *et al*., 2010), and in some areas even affected a variety of cetacean species (Mazzariol *et al*., 2018). Whereas sewage discharges have been reduced in recent years, marine aquaculture remains important in parts of the Greater North Sea (Northern Isles of Scotland), Celtic Seas (western Ireland, West Scotland) and Bay of Biscay, and may be a potential source of pathogens (OSPAR, 2010).

Removal of target and non-target species (prey depletion)

The effects of prey depletion on marine mammals are very difficult to identify directly, since species generally have varied diets and the relative importance of particular prey in the diet in relation to its abundance in the environment is poorly known. The collapse of herring stocks in the North Sea during the 1960s has been implicated in declines of harbour porpoise in the region (Evans, 1990; Reijnders, 1992), whilst during the 1990s, observed declines in porpoises in the Shetland Islands were linked to reduction in local sandeel stocks (Evans and Borges, 1995; Borges and Evans, 1997). It is possible that the continued low stock sizes of sandeels in the northern North Sea may have caused a southwards shift of porpoises since the 1990s (Hammond *et al*., 2002; 2013). Harbour porpoise is likely to be particularly vulnerable to prey depletion due to its high energetic requirements (Read and Hohn, 1995; Wisniewska *et al*., 2016). Changes in prey stocks (either by direct removal due to fishing pressure or low recruitment mediated through climate change) may have negative impacts upon their predators. Long-term declines in fish stocks have occurred in sandeel, cod and whiting in the Greater North Sea, sole and cod in the Irish Sea, and whiting in the Celtic Seas (OSPAR, 2010; 2017; ICES data download, 2018). In the Kattegat, cod stocks have also declined markedly over the last fifty years (HELCOM, 2013; 2018), whilst between 2001–2016, stocks of herring in the Gulf of Riga (Baltic Sea Region) and sprat in the western Baltic have been rated as at unfavourable status (HELCOM, 2018).

Removal of non-target species (marine mammal bycatch)

Although all marine mammal species may suffer entanglement in fishing gear, greatest concerns in the regions under review have been expressed for bycatch of harbour porpoise (Kaschner, 2003; Vinther and Larsen 2004; ICES, 2018b), short-beaked common dolphin (Tregenza *et al*., 1997; Fernández-Contreras *et al*., 2007; ICES, 2018b), minke whale (Northridge *et al*., 2010), and humpback whale (Ryan *et al*., 2016). Regions where there is strong overlap between particular gears and vulnerable species occurrence are identified as ones facing the greater threat. Gear types of particular concern for small cetaceans are bottom-set gillnets everywhere and semi-driftnets in the Baltic (harbour porpoise), pelagic or midwater trawls in the Celtic Seas and Bay of Biscay (common and striped dolphin), and creel lines in the Celtic Seas and northern part of the Greater North Sea (minke whale and humpback whale). Risso's dolphins may be prone to bycatch from longline fisheries (as reported from the Mediterranean - Macías *et al*., 2012), with longlining occurring primarily in the Celtic Seas west of Ireland south to the Bay of Biscay. Seals suffer bycatch in all regions where species occur; in Macaronesia, bycatch is a threat of concern for the highly endangered population of monk seals in Madeira (Hale *et al*., 2011).

Disturbance (e.g. wildlife watching)

Marine recreational activities including wildlife watching have increased markedly in recent decades within Europe and Macaronesia (HELCOM, 2009; O'Connor *et al*., 2009; OSPAR, 2010, 2017). There are many studies showing negative effects of recreational activities upon cetaceans or seals in the form of avoidance reactions, changes in diving behaviour, reduced feeding, etc. (e.g. Bejder and Samuels, 2003; Andersen *et al*., 2012; New *et al*., 2013; Higham *et al*., 2014; Pirotta *et al*., 2015). Species most commonly recorded as affected are the more coastal ones; bottlenose dolphin, grey seal, and harbour seal, although disturbance of other species such as minke whale (Christiansen *et al*., 2013), short-finned pilot whale (Aguilar de Soto *et al*., 2001), and Risso's dolphin (Visser *et al*., 2011) has been reported. Additionally, in Macaronesia, whale watching is reported to cause disturbance to sperm whales and bottlenose dolphins (Alves *et al*., 2018). The haul-out sites typically used by harbour seals, i.e. sandy beaches, may make them more vulnerable to disturbance due to easier human access than the, typically, more rocky areas used by grey seals.

Deliberate killing and hunting

Hunting of cetaceans does not occur within the regions under consideration. However, minke whales are taken during the Norwegian whaling activities directly to the north of the North Sea, and fin and minke whales in Icelandic waters. In the Faroes, long-finned pilot whale and to a lesser extent, Atlantic white-sided dolphin, bottlenose dolphin, Risso's dolphin, and northern bottlenose whale are taken in drive fisheries (see https://www.whaling.fo/en/regulated/450-years-of-statistics/catches/). The North Atlantic Marine Mammal Commission (NAMMCO) maintains an online database of marine mammal catches (see https://nammco.no/topics/catch-database/).

Hunting of seals occurs in Iceland, Norway, Sweden and Finland, with three species (grey, harbour, and ringed seal) taken within their range (HELCOM, 2009; 2018). Seals are often killed because they are perceived to be a threat to some human activity, for example, by taking fish of commercial value. For this reason, grey seals are taken under licence in the Baltic (HELCOM, 2009; 2018), and both grey and harbour seals in Scotland (Greater North Sea and Celtic Seas regions) under the Marine (Scotland) Act 2010 (OSPAR, 2010; Marine Scotland, 2010). Some unlicensed/illegal killing of seals almost certainly also occurs, although probably much reduced from former times, for example, of monk seals in Macaronesia (Madeira-Karamanlidis *et al.*, 2016). In the UK, it is legal to kill seals under certain conditions without licence under the Conservation of Seals Act 1970.

Deliberate killing of small cetaceans (e.g. common dolphins, bottlenose dolphins) occasionally occurs when fishers regard these as in conflict with their activities. In the latter part of the 20th century, this was quite common amongst French fishers either as bait or for human consumption (Baulaz and Morin-Repinçay, 2015). There have been no reports in recent years.

Table 4. Threat matrix for the Baltic Sea.

Baltic Sea			**HARBOUR PORPOISE**	**GREY SEAL**	**HARBOUR SEAL**	**RINGED SEAL**
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants		H	H	H	H
	Nutrient enrichment		L	L	L	L
	Microplastics		Risk of contamination leading to ill health or death possible, but no evidence of to date			
PHYSICAL LOSS	Habitat loss		L	M	L	H
PHYSICAL DAMAGE	Habitat degradation		M	M	M	H
OTHER PHYSICAL PRESSURES	Litter (including plastics and discarded fishing gear)		L	L	L	L
	Underwater noise	Military Sonar	H	L	L	L
		Seismic surveys	H	L	L	L
		Pile-driving	M	L	L	L
		Explosions	H	L	L	L
		Shipping	M	L	L	L
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)		L	L	L	L
	Death or injury by collision	Death or injury by collision (with ships)	L	L	L	L
		Death or injury by collision (with tidal devices)	Tidal devices do not exist in the region			
BIOLOGICAL PRESSURES	Introduction of microbial pathogens		L	L	L	L
	Removal of target and non-target species (prey depletion)		M	M	M	M
	Removal of non-target species (marine mammal bycatch)		H	M	M	H
	Disturbance (e.g. wildlife watching)		L	L	L	L
	Deliberate killing + hunting		Does not take place within the region	M	M	M

Table 5. Threat matrix for the Belt Seas & Kattegat.

	Belt Seas & Kattegat		**HARBOUR PORPOISE**	**GREY SEAL**	**HARBOUR SEAL**
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants		H	H	H
	Nutrient enrichment		L	L	L
	Microplastics		Risk of contamination leading to ill health or death possible, but no evidence of to date		
PHYSICAL LOSS	Habitat loss		L	L	L
PHYSICAL DAMAGE	Habitat degradation		M	M	M
OTHER PHYSICAL PRESSURES	Litter (including plastics and discarded fishing gear)		L	L	L
	Underwater noise	Military Sonar	M	L	L
		Seismic surveys	L	L	L
		Pile-driving	M	L	L
		Explosions	M	L	L
		Shipping	M	L	L
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)		L	L	L
	Death or injury by collision	Death or injury by collision (with ships)	L	L	L
		Death or injury by collision (with tidal devices)	Tidal devices do not exist in the region		
BIOLOGICAL PRESSURES	Introduction of microbial pathogens		L	L	L
	Removal of target and non-target species (prey depletion)		M	M	M
	Removal of non-target species (marine mammal bycatch)		H	M	M
	Disturbance (e.g. wildlife watching)		L	L	L
	Deliberate killing + hunting		Does not take place within the region	M	M

Table 6. Threat matrix for the Greater North Sea.

Greater North Sea			Harbour porpoise	Common dolphin	White-beaked dolphin	Atlantic white-sided dolphin	Risso's dolphin	Minke whale	Long-finned pilot whale	Killer whale	Coastal bottlenose dolphin	Grey seal	Harbour seal
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants		H	M	M	M	M	L	M	H	H	M	M
	Nutrient enrichment		L	L	L	L	L	L	L	L	L	M	M
	Microplastics		Risk of contamination leading to ill health or death possible, but no evidence to date										
PHYSICAL LOSS	Habitat loss		L	L	L	L	L	L	L	L	L	M	M
PHYSICAL DAMAGE	Habitat degradation		L	L	L	L	L	L	L	L	L	M	M
OTHER PHYSICAL PRESSURES	Litter (including plastics and discarded fishing gear)		L	L	L	L	L	M	L	L	L	M	M
	Underwater noise	Military Sonar	M	L	L	L	L	M	M	M	L	L	L
		Seismic surveys	M	L	L	L	L	M	L	L	L	L	L
		Pile-driving	M	L	L	L	L	M	L	L	M	L	M
		Explosions	M	L	L	L	L	M	L	L	M	L	M
		Shipping	M	L	L	L	L	M	L	L	M	L	L
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)		L	L	L	L	L	L	L	L	L	L	L
	Death or injury by collision	with ships	L	L	L	L	L	M	L	L	M	L	L
		with tidal devices)	Risk of collision leading to death or injury is considered possible, but no evidence to date										

Greater North Sea		Harbour porpoise	Common dolphin	White-beaked dolphin	Atlantic white-sided dolphin	Risso's dolphin	Minke whale	Long-finned pilot whale	Killer whale	Coastal bottlenose dolphin	Grey seal	Harbour seal
BIOLOGICAL PRESSURES	Introduction of microbial pathogens	L	L	L	L	L	L	L	L	L	L	M
	Removal of target and non-target species (prey depletion)	M	L	L	L	L	M	L	L	M	M	M
	Removal of non-target species (marine mammal bycatch)	H	L	L	L	L	M	L	L	L	M	M
	Disturbance (e.g. wildlife watching)	L	L	L	L	L	L	L	L	M	L	M
	Deliberate killing + hunting	Does not take place within the region									L	L

Table 7. Threat matrix for Celtic Seas including West Scotland.

Celtic Seas incl. West Scotland			HARBOUR PORPOISE	COMMON DOLPHIN	WHITE-BEAKED DOLPHIN	ATLANTIC WHITE-SIDED DOLPHIN	RISSO'S DOLPHIN	MINKE WHALE	LONG-FINNED PILOT WHALE	KILLER WHALE	FIN WHALE	SPERM WHALE	OFFSHORE BOTTLENOSE DOLPHIN	COASTAL BOTTLENOSE DOLPHIN	NORTHERN BOTTLENOSE WHALE	CUVIER'S BEAKED WHALE	SOWERBY'S BEAKED WHALE	GREY SEAL	HARBOUR SEAL
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants		H	M	M	M	M	L	M	H	L	M	M	H	L	L	L	M	M
	Nutrient enrichment		L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Microplastics		Risk of contamination leading to ill health or death possible, but no evidence to date																
PHYSICAL LOSS	Habitat loss		L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	M	M
PHYSICAL DAMAGE	Habitat degradation		L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	M	M
OTHER PHYSICAL PRESSURES	Litter (including plastics, discarded fishing gear)		L	L	L	L	L	M	L	L	L	M	L	L	L	M	M	M	M
	Underwater noise	Military Sonar	M	L	L	L	L	M	M	M	L	L	L	L	H	H	H	L	L
		Seismic surveys	M	M	M	M	M	H	M	M	H	H	M	M	H	H	H	L	L
		Pile-driving	M	L	L	L	L	L	L	L	L	L	L	M	L	L	L	M	M
		Shipping	L	L	L	L	L	M	L	L	M	L	L	L	L	L	L	L	L
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)		L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Death or injury by	with ships	L	L	L	L	L	M	L	L	M	M	L	M	L	L	L	L	L

Celtic Seas incl. West Scotland		HARBOUR PORPOISE	COMMON DOLPHIN	WHITE-BEAKED DOLPHIN	ATLANTIC WHITE-SIDED DOLPHIN	RISSO'S DOLPHIN	MINKE WHALE	LONG-FINNED PILOT WHALE	KILLER WHALE	FIN WHALE	SPERM WHALE	OFFSHORE BOTTLENOSE DOLPHIN	COASTAL BOTTLENOSE DOLPHIN	NORTHERN BOTTLENOSE WHALE	CUVIER'S BEAKED WHALE	SOWERBY'S BEAKED WHALE	GREY SEAL	HARBOUR SEAL
	collision with tidal devices	Risk of collision leading to death or injury is considered possible but no evidence to date																
BIOLOGICAL PRESSURES	Introduction of microbial pathogens	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of target and non-target species (prey depletion)	M	M	M	M	L	M	L	L	L	L	L	M	L	L	L	M	M
	Removal of non-target species (bycatch)	H	H	M	M	M	M	L	L	L	L	L	L	L	L	L	M	M
	Disturbance (e.g. wildlife watching)	L	L	L	L	L	L	L	L	L	L	L	M	L	L	L	L	M
	Deliberate killing + hunting	Does not take place within the region															M	M

Table 8. Threat matrix for The Bay of Biscay and the Iberian Peninsula.

Bay of Biscay & Iberian Peninsula			Harbour porpoise	Common dolphin	Striped dolphin	Risso's dolphin	Minke whale	Long-finned pilot whale	Killer whale	Fin whale	Sperm whale	Northern bottlenose whale	Cuvier´s beaked whale	Sowerby's beaked whale	Offshore bottlenose dolphin	Coastal bottlenose dolphin
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants		H	M	M	L	L	M	H	L	L	L	L	L	M	H
	Nutrient enrichment		L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Microplastics															
PHYSICAL LOSS	Habitat loss		L	L	L	L	L	L	L	L	L	L	L	L	L	L
PHYSICAL DAMAGE	Habitat degradation		L	L	L	L	L	L	L	L	L	L	L	L	L	L
OTHER PHYSICAL PRESSURES	Litter (including plastics and discarded fishing gear)		L	L	L	L	L	L	L	L	M	M	M	M	L	L
	Underwater noise	Sonar	L	L	L	L	M	M	L	L	L	H	H	H	L	L
		Seismic surveys	L	L	L	L	M	L	L	M	L	M	H	M	L	L
		Pile-driving														
		Shipping	L	L	L	L	M	L	L	M	L	L	L	L	L	L
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)		L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Death or injury by collision	with ships	L	L	L	L	M	L	L	H	H	L	L	L	L	L
		with tidal devices														

Bay of Biscay & Iberian Peninsula		Harbour porpoise	Common dolphin	Striped dolphin	Risso's dolphin	Minke whale	Long-finned pilot whale	Killer whale	Fin whale	Sperm whale	Northern bottlenose whale	Cuvier´s beaked whale	Sowerby's beaked whale	Offshore bottlenose dolphin	Coastal bottlenose dolphin
BIOLOGICAL PRESSURES	Introduction of microbial pathogens	L	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of target and non-target species (prey depletion)	M	L	L	L	L	L	L	L	L	L	L	L	L	M
	Removal of non-target species (bycatch)	H	H	M	L	L	L	L	L	L	L	L	L	M	M
	Disturbance (e.g. wildlife watching)	L	L	L	L	L	L	L	L	L	L	L	L	L	M
	Deliberate killing + hunting														

Table 9. Threat matrix for Macaronesia.

Macaronesia			Short-finned pilot whale	Striped dolphin	Sperm whale	Risso´s dolphin	Cuvier´s beaked whale	Blainville's beaked whale	Bottlenose dolphin	Common dolphin	Fin whale	Rough-toothed dolphin	Bryde's whale	Atlantic spotted dolphin	Monk seal
POLLUTION & OTHER CHEMICAL CHANGES	Contaminants		L	L	L	L	L	L	M	M	L	L	L	M	L
	Nutrient enrichment		L	L	L	L	L	L	L	L	L	L	L	L	L
	Microplastics		Risk of contamination leading to ill health or death possible, but no evidence to date												
PHYSICAL LOSS	Habitat loss		L	L	L	L	L	L	L	L	L	L	L	L	L
PHYSICAL DAMAGE	Habitat degradation		L	L	L	L	L	L	L	L	L	L	L	L	M
OTHER PHYSICAL PRESSURES	Litter (including plastics and discarded fishing gear)		M	L	M	M	M	M	L	L	L	L	L	L	M
	Underwater noise	Military Sonar	L	L	L	L	H	H	L	L	L	L	L	L	L
		Seismic surveys	L	L	M	L	M	M	L	L	M	L	M	L	L
		Pile-driving	No current activity but potentially harmful												
		Shipping	M	L	M	L	L	L	M	L	M	L	M	L	L
	Barrier to species movement (offshore windfarm, wave or tidal device arrays)		No current activity but potentially harmful												
	Death or injury by collision	with ships	H	L	H	L	H	L	L	L	M	L	M	L	L
		with tidal devices	No current activity but potentially harmful												

Macaronesia		Short-finned pilot whale	Striped dolphin	Sperm whale	Risso´s dolphin	Cuvier´s beaked whale	Blainville's beaked whale	Bottlenose dolphin	Common dolphin	Fin whale	Rough-toothed dolphin	Bryde's whale	Atlantic spotted dolphin	Monk seal
BIOLOGICAL PRESSURES	Introduction of microbial pathogens	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of target and non-target species (prey depletion)	L	L	L	L	L	L	L	L	L	L	L	L	L
	Removal of non-target species (bycatch)	L	L	L	L	L	L	L	L	L	L	L	L	H
	Disturbance (e.g. wildlife watching)	M	L	M	M	M	L	M	M	L	L	M	M	M
	Deliberate killing + hunting	Does not take place within the region												M

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5 ToR E. Update the database for seals

5.1 Historical context

In 2008, the WGMME recommended that a database be created for harbour and grey seal population indices within the ICES area to help ICES meet the requirements of its member countries and international organisations (e.g. OSPAR, NAMMCO, HELCOM). The recommendation was not a result of a formal request for advice from any of the above organisations, but an attempt to collate salient information to facilitate the future work of the Working Group (ICES, 2008). Despite attempts, a unified database could not be finalised and requirements with regard to seal numbers were, until 2018, met by presenting a table with only the latest counts for each area.

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 harbour (common) seal, *Phoca vitulina*, and 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 accounts for the fact that, as mobile marine predators, grey and harbour seals move across national borders. There is obvious 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 was originally focused on the Northeast Atlantic and the North Sea (relevant countries included Denmark, Germany, the Netherlands, Belgium, UK, Norway, Sweden, Belgium, France and Ireland). In later WGMME meetings, there have been discussions regarding 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), but few data points from these countries have been included before 2018.

Until 2017, WGMME's ToR A was fulfilled by presenting a table with only the latest counts for each area. In 2018, the WG discussed whether it was necessary to maintain a seal database for ICES and if the more recently collated OSPAR seal database would suffice (ICES, 2018). It was noted that data collected at the resolution used for OSPAR assessments and the European Union (EU) Marine Strategy Framework Directive (MSFD) reporting can only be produced by the bodies responsible for the collection of the data themselves. That level of resolution is not necessary for a more general inspection of trends in distribution and abundance as is done by ICES/WGMME. Secondly, it was noted that the area covered by the OSPAR seal database overlaps only partially with the area covered by the ICES/WGMME database. Also, the OSPAR database is currently updated by a formal OSPAR data call to Contracting Parties every few years; thus, data are not available annually. Finally, the OSPAR seal database currently aims at including data for assessment of the common indicators: harbour and grey seal abundance and distribution (M3) and grey seal pup production (M5). The OSPAR seal database therefore does not currently include other valuable data collected by some (but not all) parties, such as harbour seal pup production. More information on the OSPAR seal database and the common indicators are provided below.

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

- The more geographically detailed OSPAR seal database (see below) covering only EU waters will be updated 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 planned 2019 data call.
- To provide an annual overview of the status of the seal stocks concerned, the ICES/WGMME database will be maintained at a less detailed, but geographically broader level (e.g. including Iceland, Canada, USA and aiming at covering the ICES area). The database will be updated at the annual WGMME meetings (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. This WGMME database (previously 'ICES seals database') will be held and maintained by the WG.

5.2.1 Current status of the WGMME seal database

In 2018, data were collected in a Microsoft Excel spreadsheet format which sufficed to produce the figures in the report (ICES, 2018), but the tables were sensitive to accidental changes by those handling them and were not structured in a database format.

At WGMME 2019, efforts were made to restructure the data, rationalizing the area descriptions, updating the data with the most recent counts (where available), and attempting to extend the database with data from ICES regions that had not yet been incorporated.Table 1 gives an overview of the status of the data included for the different areas. At the moment, the ICES data and information collection are limited to grey and harbour seals throughout most of their range within the ICES area, as well as ringed seals in the Baltic region. For grey seals, different survey methods are used in different regions: pup counts can be used as a basis for population estimate models. Moult counts are used in some cases, as this may provide information on the trans-boundary behaviour of the animals. For the Baltic grey seals, moult counts are used as the abundance index as a large part of the pups are spread out on sea ice, complicating traditional counts. For harbour seals, most surveys reported are conducted during the moult, thus lacking data on reproduction. In the case of ringed seals, current climatic changes have caused concern with regard to the survey methods, as changing ice conditions can mean unpredictable variation of results. Moreover, ringed seals are currently included from only the Baltic area, although the species is distributed over a much larger part of the ICES area.

5.2.2 Other species and missing areas

The WG is aware that other seal species and some areas have been omitted from the former WGMME reports (

Table 1. Overview of data included in WGMME database per survey area. Years indicate coverage. Orange shading: possible data to include within the next WGMME reports. Grey shading: areas where data were merged over several years and could be included in more detail. In some areas, other seal species (ringed, harp, hooded and bearded seals and walrus) could be added in future WGMME reports; these are indicated by * in the far right column.

). This includes the Arctic species such as the ringed, harp, hooded and bearded seals and the walrus notably occurring in the more arctic zones of the ICES areas. The question is whether these all should be included in the WGMME report. WGMME is aware of the WGHARP biannual meetings and will explore if it may suffice to refer to that group's report for harp and hooded seals. For the other species and areas, either scientists working on these animals or areas could be invited to join the WGMME or reference could be made to results reported by other groups outside the ICES.

Table 1. Overview of data included in WGMME database per survey area. Years indicate coverage. Orange shading: possible data to include within the next WGMME reports. Grey shading: areas where data were merged over several years and could be included in more detail. In some areas, other seal species (ringed, harp, hooded and bearded seals and walrus) could be added in future WGMME reports; these are indicated by * in the far right column.

Survey areas			Grey seals		Harbour seals		Ringed seal
			Adults (Moult)	Pups	Adults (Moult)	Pups	Adults (Moult)
Baltic	**Western Estonia**	**Estonia**	**2003 -**				
	SW Finnish archipelago	**Finland**	**2003 -**				**2010 -**
	Gulf of Finland	**Finland, Estonia, Russia**	**2003 -**				**2010 -**
	Gulf of Riga						**1995-2018**
	Bothnian Bay and North Quark	**Sweden, Finland**	**2003 -**	**-**			**1988 -**
	Sea of Bothnia	**Sweden, Finland**	**2003 -**				
	Central Sweden	**Sweden**	**2003 -**				
	Southern Baltic	**Sweden, Denmark, Germany, Poland**	**2003 -**		**1988-2017**		
	Skagerrak	**Sweden and Denmark**					
	Kattegat/ Danish Straits	**Sweden and Denmark**			**1979-2017**		
	Southern Baltic	**Sweden and Denmark**			**1988-2017**		
	Limfjord	**Sweden and Denmark**			**1988-2017**		
	Kalmarsund	**Sweden and Denmark**			**1979-2017**		
Norway	**Svalbard**						*****
	North of 62N				**2015**		
	South of 62N				**2015**		
	Finmark				**2015**		
	Skagerrak				**1979-2016**		
	Wadden Sea	**Danish, German and**	**1980 -**	**1976 -**	**1976 -**	**1976 -**	

Survey areas			Grey seals		Harbour seals		Ringed seal
			Adults (Moult)	Pups	Adults (Moult)	Pups	Adults (Moult)
Southern North Sea		**Dutch Wadden Sea Area**					
	Delta Area	**Netherlands**	**2003 -**	**1995 -**	**1995 -**	**1995 -**	
	France		**1992 -**	**1990 -**	**1990 -**	**1990 -**	
UK	**Northern Ireland**	**Northern Ireland**	**2000-2017**	**estimate only**	**2000-2017**		
	Wales	**Wales**	**2000-2017 (estimate)**	**1977, 1992-1994, 2005 (estimate)**	**1996-2017 (estimate)**		
	England	**Southwest England**	**2000-2017 (estimate)**	**2005, 2016 (estimate)**	**1996-2017 (estimate)**		
		Northwest England	**2000-2017 (estimate)**	**-**	**1996-2017 (estimate)**		
		Northeast England	**2000-2017**	**1959-2017**	**1996-2017**		
		Southeast England	**2000-2017**	**1984-2017**	**1996-2017**		
		South England	**2000-2017 (estimate)**	**-**	**1996-2017 (estimate)**		
	Scotland	**Southwest Scotland**	**1996-2017**	**-**	**1996-2017**		
		W Scotland	**1996-2017**	**1984-2008, 2010, 2012, 2014, 2016**	**1996-2017**		
		Western Isles	**1996-2017**	**1961-2016**	**1996-2017**		
		North Coast	**1996-2017**	**1998, 2000, 2002-2008, 2014, 2016**	**1996-2017**		
		Orkney	**1996-2017**	**1960-2010, 2012, 2014, 2016**	**1996-2017**		
		Shetland	**1996-2017**	**2004**	**1996-2017**		
		E Scotland	**1996-2017**	**1997-2010, 2012, 2014, 2016**	**1996-2017**		
		Moray Firth	**1996-2017**	**2005-2009, 2012, 2014, 2016**	**1996-2017**		

Survey areas			Grey seals		Harbour seals		Ringed seal
			Adults (Moult)	Pups	Adults (Moult)	Pups	Adults (Moult)
	Ireland						
	Iceland		**1980-2016**	**1980-2016**	**1980-2016**		
	Greenland						*****
North America	**USA**			**2001-2015**			*****
	Canada	**Sable Island**		**1961- 2016**			*****
		Eastern Canada & Gulf of St Lawrence		**1996-2016**			*****
Russian Federation*							*****

5.3 The OSPAR seal database

The reporting of progress with respect to delivery of the OSPAR Strategy is based on environmental assessments undertaken by OSPAR, mainly through the various OSPAR Committees. There is a synergy between the processes of OSPAR and the MSFD and the indicator assessment process is designed in such a way that EU Member States can, if they choose, use assessment content as part of national reporting of Good Environmental Status under the MSFD (Article 8). For seals, two common indicators have been developed: M3 (harbour and grey seal population abundance and distribution) and M5 (grey seal pup production).

In 2015, OSPAR issued a formal data call to its Contracting Parties to submit data to support the assessment of these two common indicators for seals. These data formed the basis of draft assessments of indicators M3 and M5 for OSPAR's Intermediate Assessment in 2017 (OSPAR, 2017). However, some data were not submitted in the requested format, so 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 2019 data call for the purpose of completing indicator assessments for the OSPAR Quality Status Report 2023.

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).

The OSPAR seal database now refers to the collection of data generated in 2015–2016 expressly for the purpose of fulfilling the assessment criteria; this database is hosted by ICES and 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 EU, and other European Economic Area countries participating in the assessment (e.g. Norway).

The OSPAR Intermediate Assessments 2017 were performed at the scale of Assessment Units defined separately for harbour and grey seals and are summarized at the appropriate level of detail to allow assessment of abundance and distribution. The distributional aspect of the 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. Work on this database will progress in 2019 prior to the formal data call.

5.4 References

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Annex 2: Terms of Reference for 2020

2019/X/ACOMXX The **Working Group on Marine Mammal Ecology** (WGMME), chaired by Anita Gilles (Germany) and Anders Galatius (Denmark), will meet in Barcelona, Spain, 10–14 February 2020 to:

a) Review and report on any new information on seal and cetacean population abundance, distribution, population/stock structure, management frameworks (including indicators and targets for MSFD assessments), and anthropogenic threats (including cumulative effects) to individual health and population status;
b) Review foraging areas and estimate consumption by relevant seal and cetacean species in case study areas;
c) Review selected aspects of marine mammal-fishery interactions (details to be determined prior to the 2020 meeting);
d) Update the database for seals;

Justification

ToR a is a standing term of reference. However, the group proposes to expand its scope since it would be useful to include information on threats to population status, including cumulative effects of multiple stressors. Theoretical frameworks and approaches for assessing cumulative effects of multiple stressors were reviewed in 2019 (then ToR d) but new information can be provided.

ToR b aims to review species-specific foraging distributions (considering horizontal and vertical dimensions depending on data availability) and estimate consumption by marine mammal species representative in case study areas. ToR b has been agreed between WGMME and WGBIODIV to support WGBIODIV's ToR "*Investigate mechanisms linking trophic guilds under contrasting levels of pressure and/or primary production in case study areas*".

ToR c reflects common interests between WGMME and WGBYC, recognising that some aspects of marine mammal fishery interactions may otherwise not be covered by either group. As in 2019, detailed content of this ToR will be agreed between WGMME and WGBYC in consultation with the ICES Secretariat.

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

WGMME will report by 15.03.2020 for the attention of the Advisory Committee.