Cetacean Stock Assessment in North-west Europe in relation to Exploration and Production Industry Sound

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sea watch FOUNDATION

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EXECUTIVE SUMMARY

Exploration for oil and gas in North-west Europe started on any large scale in the 1960s, centred upon the North Sea. 2D seismic surveys peaked in the early 1970s before declining markedly in the late 1970s. There was a temporary increase through the 1980s and early 1990s but then became gradually less frequent. The first 3D survey was carried out in 1977 but did not become the dominant type of survey until the 1980s, reaching a peak in the middle of that decade.

Shotpoint densities were low throughout the early 1960s, and mainly in the southern North Sea, extending northwards during the late 1960s and early 1970s. During the 1970s, seismic surveys also took place throughout the Irish Sea and around Ireland, with coastal activities additionally off Spain and Portugal. Seismic activity continued in these same areas through the 1980s and 1990s, with peak shotpoint densities in the North Sea between 1985 and 1994. From 1995 onwards, seismic surveys here steadily declined, whilst starting up along the Atlantic Frontier on the edge of the continental shelf west of Britain and Ireland. However from 2000 onwards, there has been relatively little seismic survey activity anywhere in North-west Europe, with effort mostly concentrated in the same central area of the North Sea as in earlier periods.

Until the late 1970s, information on cetacean status and distribution in North-west Europe was based almost entirely upon either strandings data, or in the case of Norway, the Faroes and some Baltic States, direct catches of particular species. Sightings information began to be collected in the 1970s but effort related observations were largely confined to the coastal sector. From the 1980s onwards, effort offshore increased either from surveys targeting cetaceans or ones targeting other animal groups such as seabirds, with cetaceans being recorded at the same time. However, most effort between 1980 and 2000 has been for measures of relative rather than absolute abundance.

The first large-scale survey of absolute abundance for cetaceans over the Northwest European continental shelf was the SCANS survey undertaken in July 1994. However, it did not survey most of the Irish Sea and waters west of Britain and Ireland. A repeat survey (SCANS II) took place in July 2005, and covered a more extensive area, that included shelf seas west of Britain, in the Bay of Biscay, and around the Iberian Peninsula. Then in July 2007, a survey (CODA) was conducted covering European Atlantic waters beyond the continental shelf between 42°N and 61°N. These surveys offer an important snapshot of the numbers and distribution of different species in the month and year of that survey, although they cannot reveal trends in between surveys, on a seasonal or annual basis.

In the late 1990s, the Joint Cetacean Database (JCD) was established by the UK Joint Nature Conservation Committee in collaboration with the Sea Watch Foundation and the Sea Mammal Research Unit. The database held most effort-related cetacean sightings data available for North-west European waters, and led to the production of the *Atlas of Cetacean Distribution in North-west European Waters*, with relative abundance indices expressed as sightings per hour of observation. In the late 2000s, the JCD developed into the Joint Cetacean Protocol, with further datasets added.

The present analysis is based upon effort-based systematic survey observations (including SCANS and SCANS II, European Seabirds at Sea, and Sea Watch Foundation surveys, as well as several survey datasets from other research groups). These span a period of 30 years (1980-2009), with effort well distributed across all 5-

year time periods with greatest consistency in the North Sea, West of Scotland, and Irish Sea. Cetacean survey effort was calculated on the basis of observation hours rather than survey distance in order to integrate data from both static and mobile platforms. Corrections are made for the effect of sea state, and the resulting relative abundance indices assigned to grid cells at a resolution of 15 seconds latitude by 30 seconds longitude before plotting using ArcGIS.

Shot point densities were calculated from a database provided by IOGP by summing the estimated number of shots on a grid cell basis for each 5-year time period from 1960-2009. Maps of gridded shot-point density were then plotted using ArcGIS software.

Spatio-temporal trends were analysed for the ten most frequently observed cetacean species: harbour porpoise, bottlenose dolphin, short-beaked common dolphin, white-beaked dolphin, Atlantic white-sided dolphin, Risso's dolphin, long-finned pilot whale, killer whale, minke whale and fin whale.

In order to examine cetacean trends in relation to seismic activity for oil & gas exploration, Generalised Additive Models were run with a binomial distribution (i.e. as presence-absence models), using a forward selection based on the UBRE score. Explanatory variables were included in the following order: Shot density, Effort, Time period and Lat*Long (i.e. the interaction between Lat and Long to account for spatial autocorrelation). Two models were built for each species. For the first model, all data were used from cells with observer effort, i.e. also including sighting rates of 0. For the second model, only cells with seismic survey activity during at least one time period were included. The baseline was the time period 1980-84, and trends for each subsequent time period were then compared with that.

No cetacean species experienced a decline in sighting rates over time, whilst effort alone could not explain presence/absence per cell, and the observed temporal trends. In the first model, shot density was significant only for white-beaked (positive correlation) and common dolphin (negative correlation), when it was included in the model together with effort and time period. However, when lat*long was added to the model, shot density became non-significant and was excluded in the case of common dolphin. The presence/absence of all species was thus determined mainly by the position of the cells, and the amount of effort and time period, while shot density itself was never significant when all parameters were included. The second model yielded similar results: Shot density never reached significance in the GAMs for presence – absence except for harbour porpoise when only shot density and effort were included in the model (negative correlation). This significance became stronger when lat*long was added, but disappeared as soon as time period was included; the latter was the best model according to the UBRE score.

In both models, significant positive increases from the 1980-84 baseline were observed for the majority of time periods in minke whale, common dolphin, white-beaked dolphin, Risso's dolphin, bottlenose dolphin, and harbour porpoise. No relationship with variation in shotpoint density was found in any species.

During the last fifty years since seismic exploration started in the North Sea, there have been a number of human activities with potential impact upon cetaceans. Centuries of hunting of large whales are believed to have significantly reduced populations in the North Atlantic. Since commercial exploitation of most species ceased in the early 1980s, those stocks appear to be recovering. Minke whales continue to be hunted in Norwegian waters but in smaller numbers than formerly so their stocks may also be expected to rise.

Over the same period, there have also been major changes to the stock sizes of several commercial fish species at least in part due to over-exploitation, with a number of species experiencing significant declines during the 1960s-1970s. Whether or not overall prey depletion has occurred and caused population declines in some cetacean species is not known. However, it is likely to have influenced some cetacean distribution patterns as animals respond to regional prey shortages. The southwards shift in harbour porpoise abundance in the western North Sea between the 1990s and the present, for example, may be due to reduced sand eel stocks in the north.

Incidental mortality from entanglement in fishing gear is a well-known and worldwide problem facing cetaceans. It may have had major impacts on certain species much earlier in the twentieth century, but bycatch monitoring only started in the 1990s. During that decade it was estimated to be occurring at unsustainable levels for harbour poises in the North Sea and Celtic Sea. Since then, mortality may have declined somewhat as fisheries have become reduced although monitoring is at too low a level to establish robust bycatch estimates. There is also a bycatch of common and striped dolphins in the Celtic Sea, Bay of Biscay and around the Iberian Peninsula, whilst in northern Britain, humpback and minke whales are entangled annually in creel lines and ghost netting. However, the population impacts of bycatch mortality on any of these species are currently unclear.

The only other known cause of mortality is from vessel strikes affecting mainly sperm whale and large baleen whales such as fin whale. Shipping densities are greatest in the southernmost North Sea, Strait of Dover, English Channel, and across the Bay of Biscay. Post mortem studies in the UK indicate c. 12-20% of fin and minke whales and c. 4-5% of porpoises and dolphins can be attributed to this cause of death.

Mortality as a result of other human activities, such as by ingestion of contaminants and noise disturbance is more difficult to establish, and most impacts may be sub-lethal (though still potentially affecting populations). The first major environmental effects of persistent organic pollutants such as PCBs and pesticides like DDT and dieldrin were observed in the 1960s, with top predators such as raptorial birds most obviously affected. This led to widespread bans in the 1970s-80s, although PCBs in particular have continued to leak into the environment. Monitoring of contaminant levels in cetaceans only started routinely in the 1990s. They revealed that high PCB levels in harbour porpoise were associated with greater susceptibility to disease, and that although levels declined during the 1990s, they have remained relatively stable since. A comparison of total PCB lipid concentrations in three other European species - bottlenose dolphin, striped dolphin, and killer whale, showed that they all had levels well above the threshold normally considered to cause adverse physiological effects.

There are many sources of noise in the Northwest European marine environment. Shipping, seismic surveys, active sonar, explosions, dredging, drilling and pile driving are all sound sources suspected of causing disturbance to cetaceans. In the case of military sonar, they have been linked on occasions to actual mortality, but in general, it has not been established as yet whether population level effects may occur.

Finally, climate change may also affect the distributions of some cetacean species, with those from warmer waters extending their range northwards, presumably in response to latitudinal shifts in the range of prey species.

This study has found no evidence for a negative impact upon cetacean distributions from seismic exploratory activities in Northwest European seas. This may be because, indeed, there is no long-term impact, or that the cetacean survey data available are inadequate to demonstrate an impact; or that any impact has been masked by other strong effects. It is not possible to say which of these apply, and it may be a combination of them.

The North Sea in particular has experienced decades of seismic activity in exploration for oil and gas resources. However, whereas those seismic activities started in 1959, reaching a peak in the 1970s, dedicated cetacean surveys in the region were relatively limited until the 1990s. Thus any initial impact is unlikely to be detected. Nevertheless, if there was an initial negative impact from a large amount of seismic survey effort, it does not appear to have persisted over several decades. Species like the minke whale, bottlenose dolphin and harbour porpoise could possibly be recovering from earlier effects but there are equally plausible reasons for this being caused by other human activities (hunting in the case of the minke whale, pollution in the case of the bottlenose dolphin, and both bycatch and pollution in the case of the harbour porpoise).

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1. INTRODUCTION

1.1 Objectives

The general aim of this project is to review the available information in order to assess whether it is possible to determine if the offshore EandP industry has had any long-term influence upon status changes in cetacean stocks within North-west Europe.

1.2 Specific Aims

i. Using available cetacean stock data, examine the relationships between EandP industry operation sounds and cetacean stock trends. This requires assessing whether existing data will allow meaningful analysis, and reviewing the literature relating to interaction of cetacean stocks with EandP activity.
ii. Review the current status and trends of different cetacean stocks that are potentially exposed to sound generated by the oil and gas industry in the marine environment.
iii. Examine the extent to which status changes occur following major anthropogenic influences (e.g. whaling) and differ between species; and then to determine how this relates to sound exposures, particularly for stocks whose habitats are spatially relevant to the EandP industry.
iv. Identify factors that are key to controlling or influencing cetacean population growth rates of various stocks (e.g. anthropogenic sound, by-catch, whale watching, climate change, etc).
v. Determine whether there are key species or regions that would lend themselves to more detailed analyses or data collection and if so, what species, analyses or data collection would be appropriate.

1.3 Study Area

In the context of this project, the seas around North-west Europe are taken as equivalent to the ASCOBANS Agreement Area. This is defined as "the marine environment of the Baltic and North Seas and contiguous area of the North East Atlantic, as delimited by the shores of the Gulfs of Bothnia and Finland; to the south-east by latitude 36°N, where this line of latitude meets the line joining the lighthouses of Cape St. Vincent (Portugal) and Casablanca (Morocco); to the south-west by latitude 36°N and longitude 15°W; to the north-west by longitude 15° and a line drawn through the following points: latitude 59°N/longitude 15°W, latitude 60°N/longitude 05°W, latitude, 61°N/longitude 4°W;latitude 62°N/ longitude 3°W; to the north by latitude 62°N; and including the Kattegat and the Sound and Belt passages" (www.ascobans.org/the_agreement.html).

1.4 Cetacean Fauna

Thirty-five species of cetaceans have been recorded in North-west Europe, although many of these normally live outside the region and are therefore recorded as vagrants. A full list of species and their Latin names are given in Appendix 1. Sixteen of those species may be regarded as native to the region (indicated in bold in Appendix 1). The status of each species by country is summarised in Appendix 2.

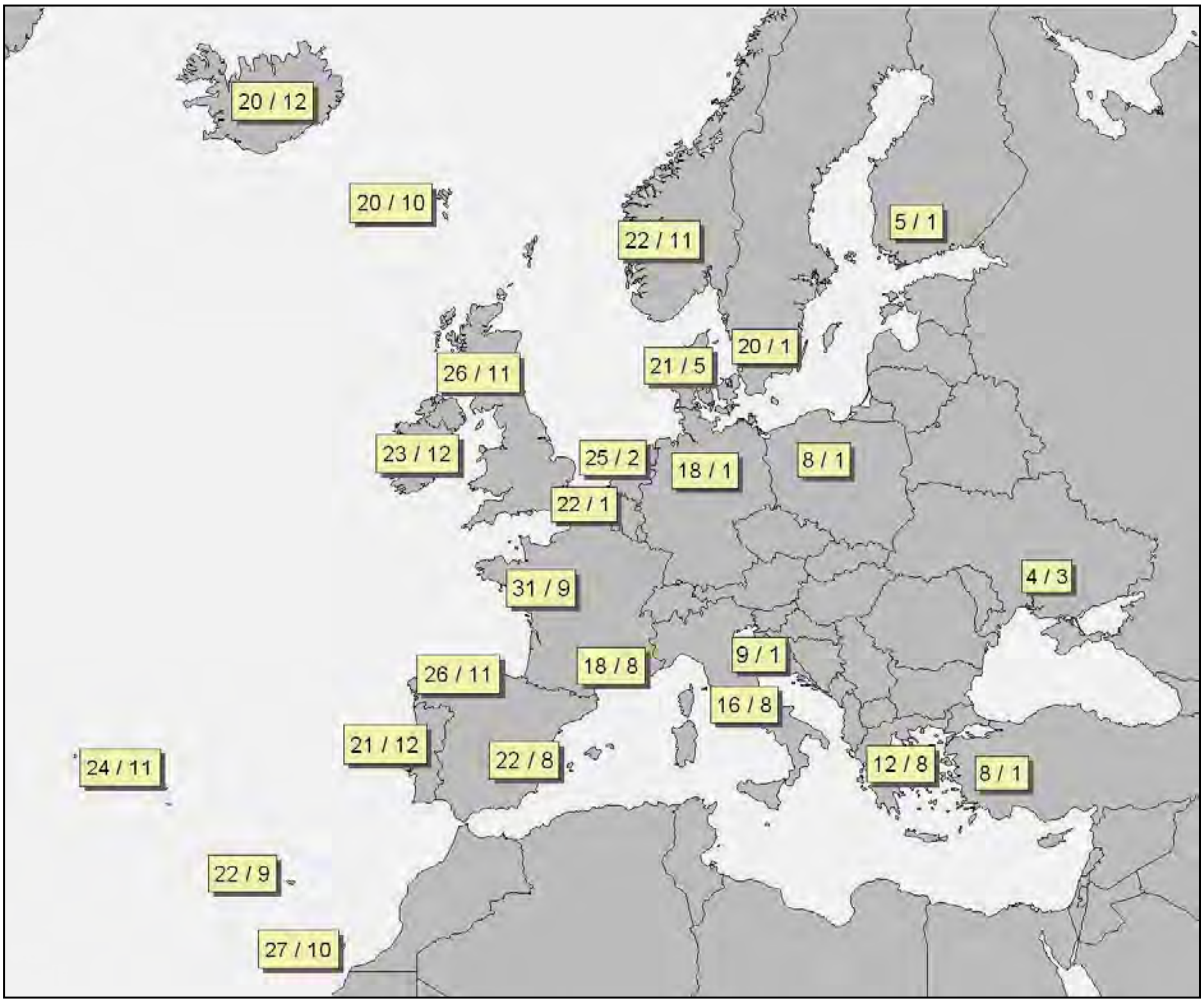


Fig. 1. Map of Cetacean Species Diversity in Europe by Country
(The first value relates to the total number of species recorded in that country;
the second value relates to the number of species occurring there regularly)

Within Europe, there is a trend in species diversity from east to west, reflecting the influence of the North Atlantic (Fig. 1). Thus, the cetacean faunas of a country and particularly the number of species occurring regularly are highest in those countries bordering the Atlantic, and are lowest in the Baltic and Black Seas, and the Eastern Mediterranean. Understanding regional effects upon cetaceans of anthropogenic activities needs to take this into consideration.

2. METHODS

2.1 Cetacean Data Sources

Until the late 1970s, information on cetacean status and distribution in North-west Europe was based almost entirely upon either strandings data, or in the case of Norway, the Faroes and some Baltic States, direct catches of particular species (Table 1).

Table 1. Sources of Information from strandings and catch data, 1900-2015

Country	Literature Source
Norway	Christensen & Ugland, 1984; Øien, 1988
Sweden	Lindstedt & Lindstedt, 1988, 1989; Berggren, 1994, 1996; Carlstrom, 2003
Denmark	Degerbøl, 1935; Bondesen, 1951, 1977; Lowry & Teilmann, 1994; Kinze *et al.,* 1997; Kinze, 1995a,b, 2006, 2011; Kinze *et al*., 2000, 2001, 2003; Kinze & Jensen, 2001; Vinther & Larsen, 2002, 2004; Lockyer & Kinze, 2003; Jensen *et al*., 2009; Kinze *et al*., 1998, 2010
Finland	Määttänen, 1990; Kujala, 2006; Coalition Clean Baltic, 2006
Poland	Ropelewski, 1952a, b; Jakuczsen, 1973; Skóra, 1991; Coalition Clean Baltic, 2006; Kuklik, 2007
Germany	Weber, 1922; Mohr, 1935; Schultz, 1970; Goethe, 1983; Kremer, 1987; Schulze, 1991, 1996; Bohlken *et al*., 1993; Benke *et al*., 1998; Kölmel & Wurche, 1998; Siebert *et al*., 2001; Hasselmeier *et al*., 2004; Siebert *et al*., 2006
The Netherlands	Weber, 1922; Van Deinse, 1925, 1931; 1944-66; Stopelaar *et al*., 1935; Utrecht & Husson, 1968; Husson & Van Bree, 1972, 1979; Van Bree & Husson, 1974; Van Bree, 1977; Van Bree & Smeenk, 1978, 1982; Smeenk, 1986, 1987, 1989, 1992, 1995, 2003; Bakker & Smeenk, 1990; Smenk *et al*., 1994; Addink & Smeenk, 1999; Kompanje, 2001, 2005; Camphuysen & Peet, 2006; Leopold & Camphuysen, 2006; Camphuysen & Oosterbaan, 2009; Keijl *et al*., 2015
Belgium	De Smet, 1974, 1978, 1981; Van Gompel, 1991, 1996; Haelters & Camphuysen, 2009; Jauniaux *et al*., 2008
France (Atlantic)	Duguy, 1972-92; Duguy & Hussenot, 1982; Collet *et al*., 1999; Collet & Van Canneyt, 1999; Van Canneyt, 2000-02, 2005; Van Canneyt & Dorémus, 2003; Van Canneyt & Peltier, 2006; Van Canneyt & Chauvel, 2007; Van Canneyt *et al*., 1998a, b, 1999, 2000, 2004, 2008, 2009, 2010, 2014; Dabin *et al*., 2011; Peltier *et al*., 2013, 2014; Authier *et al*., 2014
United Kingdom	Harmer, 1914-27; Fraser, 1934, 1946, 1953, 1974; Sheldrick, 1976, 1989; Evans, 1980, 1992; Sheldrick *et al*., 1992, 1994; Kuiken *et al*., 1994; Evans, 1997; Kirkwood *et al*., 1997; Bennett *et al*., 2000; Muir *et al*., 2000; SAC, 2000; Evans *et al*., 2003; Sabin *et al*., 2003-06; MacLeod *et al*., 2005; Canning *et al*., 2008; Deaville & Jepson, 2007-09, 2011
Republic of Ireland	O'Riordan, 1972, 1981; Fairley, 1981; Berrow & Rogan, 1997; O'Brien *et al*., 2009; O'Connell & Berrow, 2010
Spain (Atlantic)	Cabrera, 1914; Casinos & Vericad, 1976; Penas-Patiño & Seage, 1989; Cendrero, 1993; López *et al*., 2002
Portugal	Teixeira, 1979; Reiner, 1985; Penas-Patiño & Seage, 1989; Sequeira *et al*., 1992, 1996; Silva & Sequeira, 2003; Brito *et al*, 2009

Strandings schemes have long been in place in the Netherlands, Belgium, United Kingdom and France, with reporting schemes established more recently in Denmark, Sweden, Germany, Ireland, Spain and Portugal. Elsewhere, strandings reports tend to be incidental.

Sightings information began to be collected in the 1970s but effort related observations were largely confined to the coastal sector (Verwey, 1975; Evans, 1976, 1980). From the 1980s onwards, effort offshore increased either from surveys targeting cetaceans or ones targeting other animal groups such as seabirds, with cetaceans being recorded at the same time (see Table 2 for a list of the main publications relating to sightings surveys).

Table 2. Sources of Information from sightings data, 1900-2015

Country	Literature Source
Norway	Bjørge & Øien, 1995; Øien, 1989, 1991, 1995, 1999, 2005, 2009, 2010; Isaksen and Syvertsen, 2012
Sweden	Berggren, 1994, 1996; Berggren & Arrhenius, 1995a, b; Hiby & Lovell, 1996; Berggren *et al*., 2004; Teilmann *et al*., 2008
Denmark	Kinze, 1984; Kinze & Sørensen, 1984; Heide-Jørgensen *et al*., 1992, 1993; Skov *et al*., 1994, 1995; Kinze, 1995a, b; Kinze *et al*., 1997; Teilmann & Lowry, 1996; Bloch, 1998; Lockyer & Kinze, 2003; Tougaard *et al*., 2006a, b; Teilmann *et al*., 2008; Skov & Parnas, 2009; Sveegaard *et al*., 2011, 2012
Finland	Gillespie *et al*., 2005; Kujala, 2006
Poland	Skóra *et al*., 1988; Skóra, 1991; Skóra & Kuklik, 2003; Berggren *et al*., 2004; Gillespie *et al*., 2005
Germany	Heide-Jørgensen *et al*., 1992, 1993; Benke *et al*., 1998; Siebert *et al*., 2006; Berggren *et al*., 2004; Scheidat *et al*., 2004, 2008; Rye *et al*., 2008; Scheidat & Verdaat, 2009; Gilles *et al*., 2009, 2011, 2012a, b, 2014a, b; Peschko *et al*., 2016
The Netherlands	Verwey, 1975; Camphuysen, 1982; Camphuysen & Van Dijk, 1983; Baptist, 1987; Camphuysen & Leopold, 1993; Camphuysen, 1994, 2004; Baptist & Witte, 1996; Witte *et al*., 1998; Osinga, 2005; Camphuysen & Peet, 2006; Van der Meij & Camphuysen, 2006; Camphuysen & Heijboer, 2008; Haelters & Camphuysen, 2009; Scheidat & Verdaat, 2009; Scheidat *et al*., 2012; Geelhoed *et al*., 2014; Peschko *et al*., 2016
Belgium	Van Gompel, 1991, 1996; Courtens *et al*., 2008; Haelters & Camphuysen, 2009; Haelters *et al*., 2011
France (Atlantic)	Liret, 2001; Kiszka *et al*., 2004, 2007; Liret *et al*., 2006; Certain *et al*., 2008; Ricart *et al*., 2014; Pettex *et al*., 2014; Louis & Ridoux, 2015; Louis *et al*., 2015; Couet, 2015
United Kingdom	Evans, 1976, 1980, 1981, 1988, 1992; Blake *et al*., 1984; Evans *et al*., 1986, 2003; Tasker *et al*., 1987; Webb *et al*., 1990; Tregenza, 1992; Northridge *et al*., 1995, 1997; Bloor *et al*., 1996; Stone, 1997, 1998, 2000, 2001, 2003a, b, 2006; Pollock *et al*., 1997, 2000; Williams *et al*., 1997; Wilson *et al*., 1997, 1999, 2004; Weir *et al*., 2001; MacLeod *et al*., 2003, 2004; 2006; MacLeod, 2004; MacLeod *et al*., 2005, 2007; Liret *et al*., 2006; Weir *et al*., 2007; Robinson *et al*., 2007, 2009; Tetley *et al*., 2008; Pesante *et al*., 2008; Corkrey *et al*., 2008; Baines & Evans, 2009, 2012; Pierpoint *et al*., 2009; Bolt *et al*., 2009; Marubini *et al*., 2009; Embling *et al*., 2010; Pikesley *et al*, 2011; Anderwald *et al*., 2012a; Isojunno *et al*., 2012; Booth *et al*., 2013; Cheney *et al*., 2013, 2014; De Boer *et al*., 2014; McClellan *et al*., 2014; Feingold & Evans, 2014a, b; Evans *et al*., 2015; Heinanen & Skov, 2015; Norrman *et al*., 2015; Paxton *et al*., 2016
Republic of Ireland	Evans, 1981; Leopold *et al*., 1992; Leopold & Couperus, 1995; Rogan & Berrow, 1996; Berrow *et al*., 1996, 2002; Pollock *et al*., 1997; Rogan *et al*., 2000; Ingram, 2000; Ingram *et al*., 2001, 2003; Ingram & Rogan, 2003; O'Cadhla *et al*., 2003; Englund *et al*., 2007, 2008; O'Brien *et al*., 2009; Berrow *et al*., 2009, 2010a, b; Ingram *et al*., 2009; Berrow *et al*., 2012; Anderwald *et al*., 2012b; Wall *et al*., 2013; O'Brien & Berrow, 2014; Cronin & Barton, 2015; Rogan *et al*., 2015; Nykanen *et al*., 2015
Spain (Atlantic)	Sanpera & Jover, 1989; Penas-Patiño & Seage, 1989; López *et al*., 2004; Pierce *et al*., 2010; López *et al*., 2012, 2013; Goetz *et al*., 2014
Portugal	Dos Santos & Lacerda, 1987; Harzen, 1998; Gaspar, 2003; Brito *et al*., 2009; Silva *et al*., 2009; Araújo *et al*., 2014; Vingada, 2012; Goetz *et al*., 2014; Martinho *et al*., 2015; Correia *et al*., 2015; Lacey, 2015
International	Hammond *et al*., 1995, 2002, 2013; Reid *et al*., 2003; Hammond, 2008; MacLeod & Hammond, 2008; Lockyer & Pike, 2009; CODA, 2009; Hammond *et al*., 2013; Murphy *et al*., 2013; Gilles *et al*., 2016

Table 2 clearly shows that the growth in information about cetacean status and distribution from sightings surveys is very recent, with 70% of publications occurring since the year 2000. This poses challenges when one is attempting to identify trends over longer time periods and trying to assess the possible impacts of different human activities including oil and gas explorations. With very little offshore survey effort before 1980, this was taken as the threshold for spatio-temporal trends. Most effort between 1980 and 2000 has been for measures of relative rather than absolute abundance.

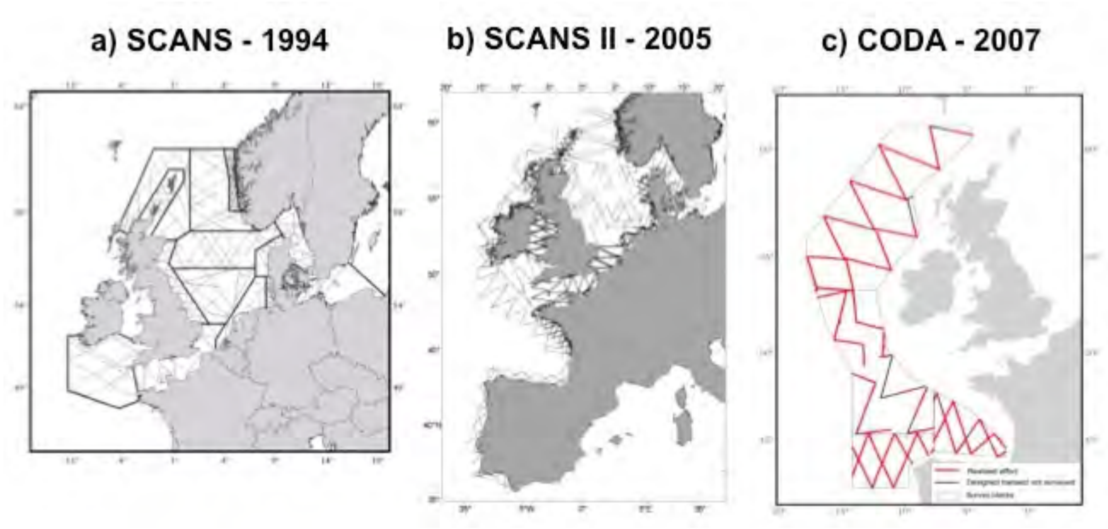


Fig. 2. Maps of line-transects undertaken during a) SCANS; b) SCANS II; and c) CODA surveys

The first large-scale survey of absolute abundance for cetaceans over the Northwest European continental shelf was the SCANS survey undertaken in July 1994 (Hammond *et al*., 1995, 2002). Although covering all of the North Sea, Celtic Sea, English Channel, Skagerrak, Kattegat, inner Danish waters and the western Baltic, (Fig. 2a). A repeat survey (SCANS II) took place in July 2005, and covered a more extensive area, that included shelf seas west of Britain, in the Bay of Biscay, and around the Iberian Peninsula (Hammond *et al*., 2013; Fig. 2b). Then in July 2007, a survey (CODA) was conducted covering European Atlantic waters beyond the continental shelf between 42°N and 61°N (CODA, 2009; Fig. 2c). These surveys offer an important snapshot of the numbers and distribution of different species in the month and year of that survey, although they cannot reveal trends in between surveys, on a seasonal or annual basis.

In the late 1990s, the Joint Cetacean Database (JCD) was established by the UK Joint Nature Conservation Committee (JNCC), working in collaboration with the Sea Watch Foundation (SWF) and the Sea Mammal Research Unit (SMRU). The database held most effort-related cetacean sightings data available for North-west European waters, and led to the production of the *Atlas of Cetacean Distribution in North-west European Waters* (Reid *et al*., 2003), with relative abundance indices expressed as sightings per hour of observation.

In the late 2000s, the JCD developed into the Joint Cetacean Protocol (JCP), with further datasets added. The present analysis is based upon effort-based systematic survey observations (including SCANS and SCANS II, European Seabirds at Sea, and Sea Watch Foundation surveys, as well as several survey datasets from other research groups). These span a period of 30 years (1980-2009), with effort well distributed across all time periods (see section 3.1). Detailed descriptions of data sources can be found in Evans and Wang (2003), Reid *et al*. (2003), and Paxton *et al.* (2016).

2.2 Data Treatment

2.2.1 Treatment of Study Area

As noted in section 1.3, the study area comprised the Baltic and North Seas, and the North-east Atlantic between 65° and 36° North, with a western boundary at 20° West. This broadly equated to the original ASCOBANS Agreement Area but very little data exist for the Baltic Proper so a line was drawn at the eastern edge of Danish waters. A grid was then laid over the study area, each cell in the grid measuring 15 seconds latitude by 30 seconds longitude. All data were assigned to their respective grid cells.

2.2.2 Cetacean Sightings Data

Cetacean sightings data were sourced from the Sea Watch Foundation database, European Seabirds at Sea (ESAS) vessel data, SCANS and SCANS II aerial and vessel data, and CODA vessel data (Evans *et al*., 2003; Reid *et al*., 2003; Hammond *et al*., 1995, 2002, 2013; CODA, 2008). Other data sets including those from marine mammal observers placed on seismic survey vessels were not included in order to reduce heterogeneity in survey procedures.

Cetacean data were compiled in an Access database holding tables for survey effort and sightings. The data were organised by assigning each record to a grid cell and, where necessary, effort data were split into segments at cell boundaries.

In order to examine long-term status changes of cetacean species in relation to oil and gas exploration in this region, one is forced to use measures of relative abundance rather than absolute densities. These are normally expressed in terms of numbers of individuals of a particular species per time of observation or km travelled.

Cetacean survey effort was calculated on the basis of observation hours rather than survey distance in order to integrate data from both static and mobile platforms. Survey hours were summed for each grid cell and maps showing the distribution of survey effort were plotted for each time period.

Table 3. Correction factors applied, according to sea state

	Sea State					
Species	**0**	**1**	**2**	**3**	**4**	**>4**
Harbour Porpoise	1.00	0.57	0.34	0.14	0.14	0.04
Dolphins	1.00	1.10	0.93	0.61	0.41	0.41
Minke whale	1.00	0.97	0.75	0.25	0.22	0.13
Large whales	1.00	1.00	1.00	1.00	1.00	1.00

Preliminary examination of the data indicated that the single factor most influencing sighting rates of any species in any survey, was sea state. In general, surveys are conducted only when sea state is Beaufort scale 2 or less, and in good visibility, but on occasions conditions can worsen during the course of a survey. Therefore in order to calculate comparable sighting rates, the effort data were first corrected for the effect of sea state for each of the following four species groups: harbour porpoise, dolphin species, minke whales, and large whales. This was achieved by calculating the overall sightings rates in each sea state category and dividing each rate by the rate at sea state 0. The correction factors so derived were used to scale the effort before calculating the number of sightings per corrected hour of observation (Table 3).

In the case of minke whales and harbour porpoises the rates calculated were of the number of individual animals seen per hour, while for all other species the number of sightings per hour was calculated, i.e. a sighting of a group of any number of animals was considered to be a single sighting.

Plots of sighting rates were interpolated using the Inverse Distance Weighted method in ArcGIS in order to generate smoothed maps of cetacean distribution. Input points were calculated as the mean position of sightings for any given species within each cell, rather than the cell centroid. Low levels of effort in some cells can give rise to unreasonably high sightings rates, and interpolation may effectively spread such spuriously high values into neighbouring areas. For this reason, data from cells with low levels of effort (2 hours or less per cell) were filtered out before applying the interpolation process.

2.2.3 Seismic Survey Data

A dataset was supplied by IOGP, holding details of all seismic surveys carried out worldwide up to the end of 2008. All offshore seismic surveys within the study area were extracted and a database compiled including the following data for each survey: the year the survey commenced; the type of survey (2D, 3D, 4D); the total number of km surveyed (for 2D surveys) or the survey area in km^2 (for 3D and 4D surveys); and the co-ordinates of the mid point of the survey. In some cases data for the length or area of the survey were unavailable, in which case the mean value for the respective survey type was used.

For the purposes of this study there is no difference between a 4D and 3D survey; a 4D survey simply being a repeated 3D survey. Both 2D and 3D surveys use a similar source, implemented using a towed array of airguns, the main difference between them in terms of potential impacts on marine mammals being the survey design and the implication this has on the density of shots fired. 2D surveys tend to be linear with wide spacing between sail lines, while 3D surveys are designed such that the entire survey area is covered by a regular array of closely spaced lines. The interval between consecutive shots is similar for the two types of survey, typically around 25m, and on that basis a value of 40 shots per km was assumed for 2D surveys and 800 shots per km^2 for 3D and 4D surveys.

The only position data available were the centre points of each survey; no information was given on the extent of individual survey areas. Each survey was assigned to a single grid cell, but it should be borne in mind that this is an approximation as many surveys are likely to have extended beyond the boundaries of the cell within which their mid point was located. Shot point densities were calculated by summing the estimated number of shots in each grid cell for each time period. Maps of gridded shot-point density were then plotted using ArcGIS software.

3.0 RESULTS

3.1 Survey Effort

Interpolated maps of relative abundance were produced for cetaceans across the entire ASCOBANS Agreement Area. A number of dedicated surveys using similar field protocols have been undertaken in the thirty years between 1980 and 2010 in Northwest European waters. Data from several of those surveys (including the 1994 and 2005 synoptic SCANS abundance surveys, the CODA shelf edge survey in 2007, and the ESAS and SWF databases were collated, corrected for effort, and the effects of sea state upon detection rates incorporated (although c. 75% of effort was in sea states of 2 or less – see Fig. 3). Relative abundance was plotted using interpolation by inverse distance weighting according to the procedures described in section 2.2.

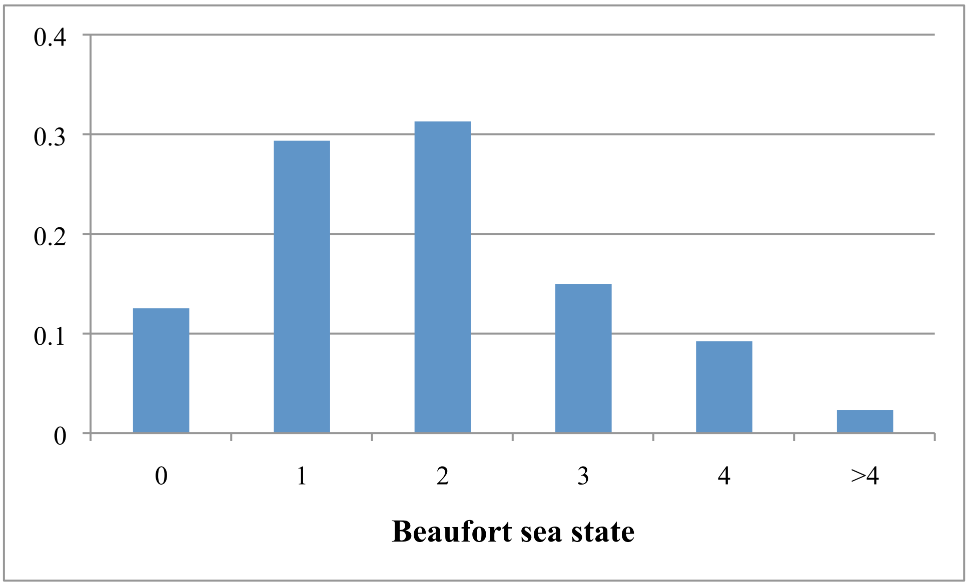


Fig. 3. Proportion of Effort at different Sea States

Most surveys were conducted during summer months (May – October), effort being lowest between December and March.

Table 4. Distribution of Effort by Survey/Platform Type

Effort type	Hours	Percent
Ferry	1629.33	1.63
General vessel surveys	15035.18	15.04
ESAS	45525.23	45.55
Line transect surveys	3885.52	3.89
Static	30905.17	30.92
Aerial	2969.25	2.97
TOTAL	99950.68	100.00

The database used spanned thirty years of survey effort, totalling just under 100,000 hours (see Table 4), and yielding 44,500 sightings.

The proportion of effort by time period is shown in Fig. 4.

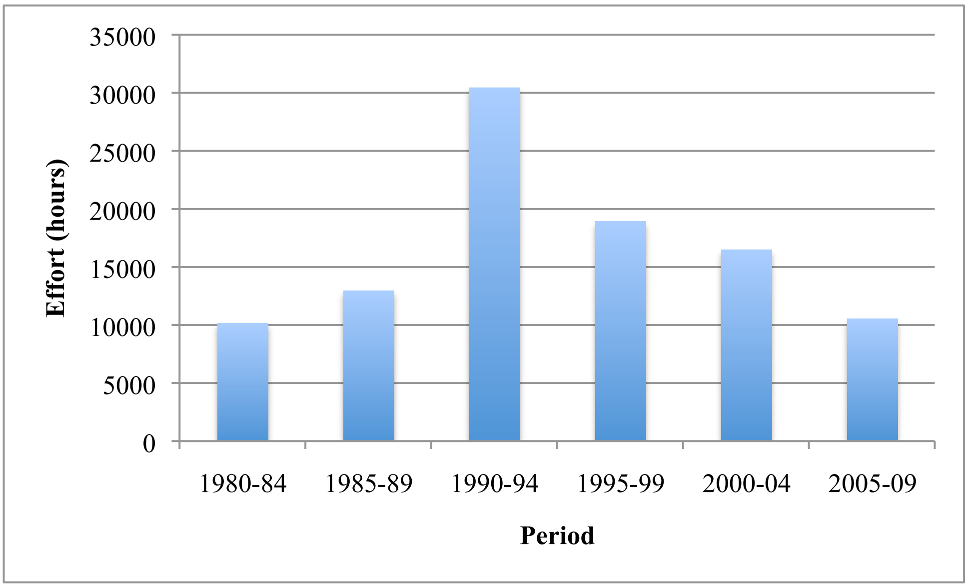


Fig. 4. Proportion of Effort by Time Period

Survey effort available for analysis has varied across time periods, being greatest in the 1990s and early 2000s. Spatial coverage has also varied to an extent although in general it has been not changed substantially between time periods, with greatest consistency in the North Sea, West of Scotland, and Irish Sea (Fig. 5a, b).

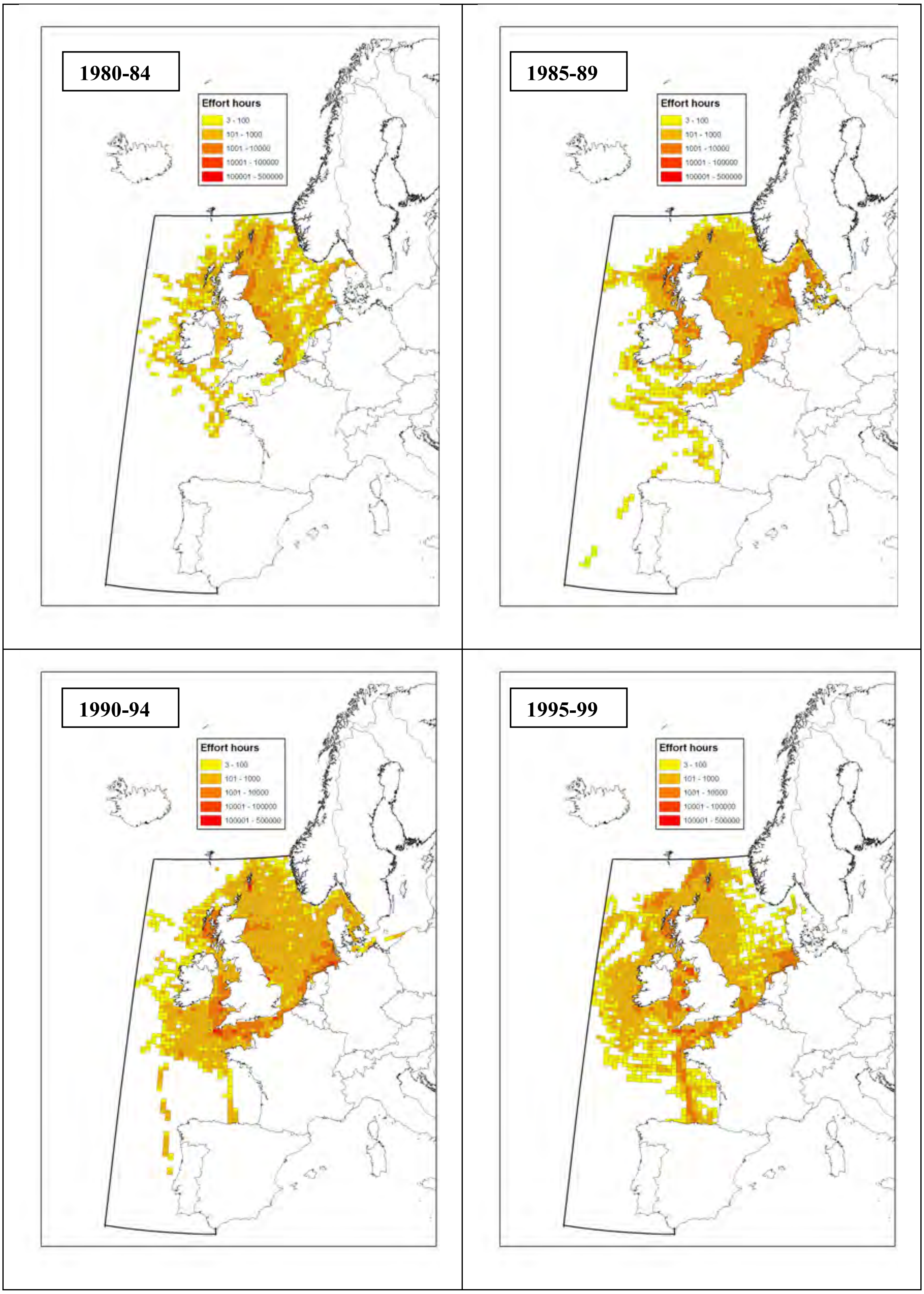


Fig. 5a. Distribution of Survey Effort, 1980-99

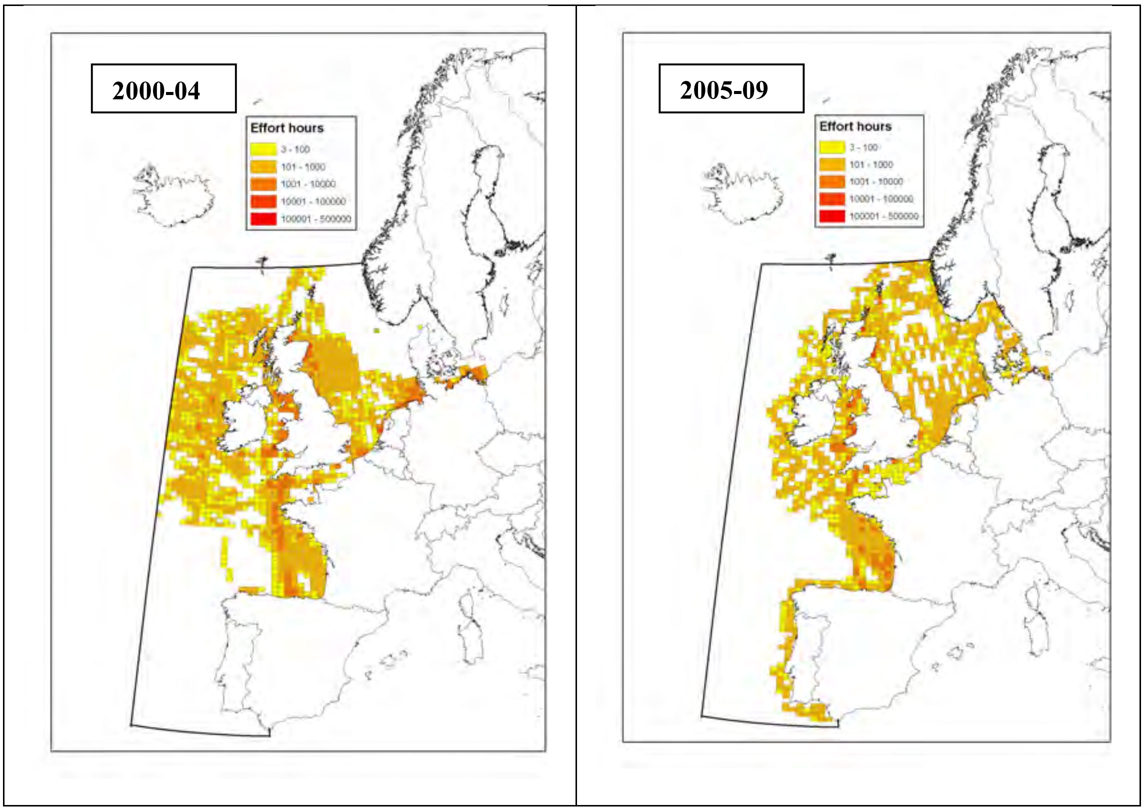


Fig. 5b. Distribution of Survey Effort, 2000-09

3.2 Spatio-temporal Trends for Major Cetacean Species

3.2.1 Overall Distributions

The most abundant and widely distributed cetacean species in Northwest European shelf seas is the **harbour porpoise** (Fig. 6a). This has been demonstrated repeatedly in a variety of studies (see, for example, Hammond *et al*., 1995, 2002, 2013; Evans *et al*., 2003; Reid *et al*., 2003). Concentrations occur in several areas (e.g. western North Sea, Northern Isles, Inner Hebrides, West Wales, and Southwest Ireland) but the species is relatively uncommon south of the British Isles. Other studies have indicated a re-distribution of the species in the western North Sea, with increased abundance in the southern North Sea and eastern English Channel and a decrease in the northwestern North Sea (Evans *et al*., 2003; Kiszka *et al*., 2004, 2007; Haelters and Camphuysen, 2009; Hammond *et al*., 2013).

Bottlenose dolphins have a predominantly Atlantic distribution with greatest concentrations along the shelf edge, although coastal populations exist in scattered locations (Fig. 6a). In the North Sea, the species is largely confined to coastal waters of Northeast Britain.

The **short-beaked common dolphin** is also largely absent from the North Sea, occurring mainly in the northwest sector. It has an Atlantic distribution with highest concentrations in the south around the Iberian Peninsula, and in the Celtic Sea south of Ireland, Southwest Wales and Southwest England (Fig. 6a). Largest numbers occur in offshore waters beyond the continental shelf.

The distribution of **white-beaked dolphin** is centred upon the northern North Sea and in shelf seas northwest of the British Isles (Fig. 6a). The species is rare south of Britain and Ireland.

Atlantic white-sided dolphins are distributed mainly in the north of the region, with concentrations to the north and west of the British Isles beyond the shelf edge, although the species also occurs in the northwest sector of the North Sea (Fig. 6b). The species is largely absent from the Irish Sea and is a vagrant south of the British Isles.

Risso's dolphin also has a predominantly Atlantic distribution, scarcely entering the North Sea (Fig. xx). Nowhere is it common although there are localised areas where the species is recorded on a regular basis (Fig. 6b).

The **long-finned pilot whale** is much more common than the Risso's dolphin, but very much concentrated in waters along the shelf break west of the British Isles and in the Bay of Biscay (Fig. 6b). It is rare in most of the North Sea but is recorded in the western sector of the English Channel.

Killer whale is for the most part restricted to deep waters north of the British Isles although it enters shelf seas around Shetland and Orkney, the Hebrides, and Western Ireland (Fig. 6b). The species is patchily distributed in the Bay of Biscay, being rare south of the British Isles.

The **minke whale** s widely distributed over the Northwest European shelf, with concentrations in the central North Sea, west of Scotland, and in the Celtic Sea (Fig. 6c). The species occurs also in the eastern part of the Bay of Biscay but is rare or absent around much of the Iberian Peninsula.

The **fin whale** is also widely distributed but, unlike minke whale, it is much less common and recorded mainly far west of Atlantic coasts beyond the continental shelf edge (Fig. 6c). Concentrations of the species occur mainly in the central Bay of Biscay.

3.2.2 Temporal Trends

The distribution plots by 5-year time period suggest that **harbour porpoises** were relatively uncommon during the 1980s and early 1990s (Fig. 7a). This is indicated also by other studies (Evans *et al*., 2003; Paxton *et al*., 2016), and may be a legacy from earlier periods when general declines were noted (Evans, 1980, 1990, 1992; Smeenk, 1987; Reijnders, 1992). After the mid-1990s, the species seems to have increased particularly in the southern North Sea and in the Celtic Sea (Fig. 7a, b).

The more localised distribution of **bottlenose dolphin** makes it more difficult to interpret temporal trends since some areas where the species traditionally occurs have not been surveyed consistently over the 30-year period. Thus Cardigan Bay in West Wales, which hosts a long-term bottlenose dolphin population, was not surveyed during the early 1980s, and the offshore population was not well sampled until the mid-1990s. From that period onwards, the species was consistently found offshore west of Ireland and in the Bay of Biscay (Fig. 8a, b).

The relatively low offshore coverage in the 1980s means that temporal trends in **short-beaked common dolphin** distribution can only really be examined from the 1990s onwards (Fig. 9a, b). Those show the same general areas occupied in each five-year period, with little indication of a distributional shift over the study period.

Survey coverage in the North Sea has been reasonably consistent across time periods. This allows for a better assessment of temporal trends in the distribution of the **white-beaked dolphin** given its largely North Sea distribution. This indicates a distributional shift northwards out of the southernmost North Sea from the mid-1990s onwards but with little change since then (Fig. 10a, b).

The **Atlantic white-sided dolphin** appears to have largely moved out of the North Sea in the latest time period, being recorded only in the northernmost part around the Shetland Isles (Fig. 11a, b). It has maintained its shelf edge distribution west of Britain and Ireland.

The Atlantic distribution of **Risso's dolphin** is maintained across time periods, with some incursions into the northwestern North Sea, during the 1990s and late 2000s (Fig. 12a, b).

As with the short-beaked common dolphin, the low survey effort offshore during the 1980s makes it difficult to assess distribution trends for a pelagic species like **long-finned pilot whale.** However, there is no indication of a distributional shift, and over the entire thirty year period, the species is largely absent from the North Sea (Fig. 13a, b).

Across the time periods, sighting rates of **killer whale** are uncommon and concentrated around the northern perimeter of the North Sea and the Hebrides of Scotland, with no indication of a distribution shift or obvious overall decline (Fig. 14a, b).

Sighting rates of **minke whale**, on the other hand, appear to have increased and extended their range between the 1980s and the 1990s, and since then have been maintained (Fig. 15a, b). A similar finding has been found in some other studies (Evans *et al*., 2003; Paxton *et al*., 2016) whilst a comparison of abundance estimates in the North Sea between July 1994 (SCANS survey) and July 2005 (SCANS II survey) indicated an increase from 7,250 to 10,786, although non-significant (Hammond *et al*., 2013). Other survey estimates in the North Sea indicated high numbers during the mid to late 1990s (20,294 in 1995, and 11,713 in 1998) (Schweder *et al*., 1997; Skaug *et al.,* 2004; Bøthun *et al*., 2009), suggesting that there could be large inter-annual variations.

There is very little offshore survey effort during the 1980s, and so temporal comparisons of **fin whale** distributions start in 1990. Sighting rates since then show a predominantly Atlantic distribution with no indication of a distributional change between 1995 and 2009 (Fig. 16).

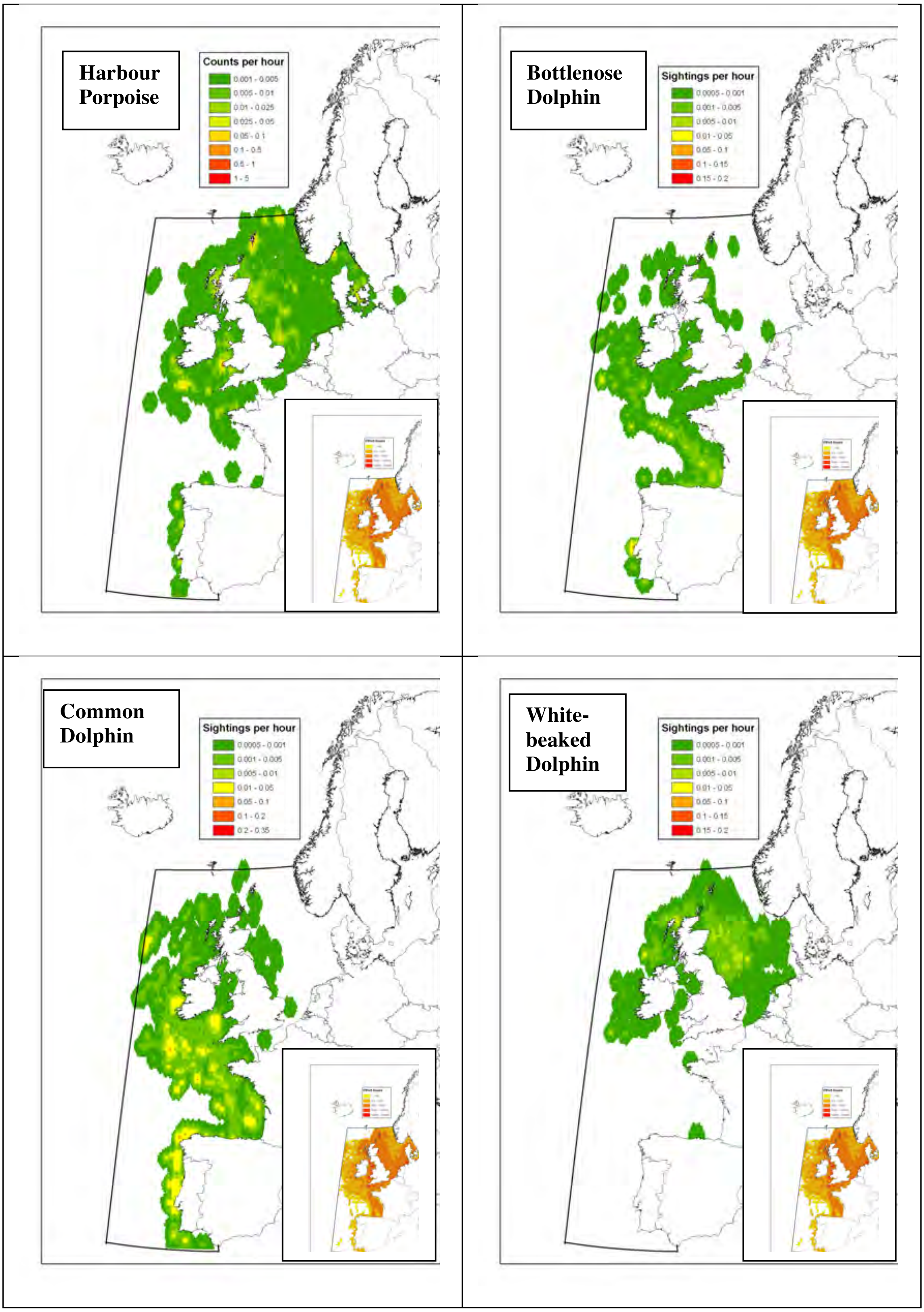


Fig. 6a. Overall Distributions of Cetaceans, 1980-2009
(Inset shows Distribution of Effort)

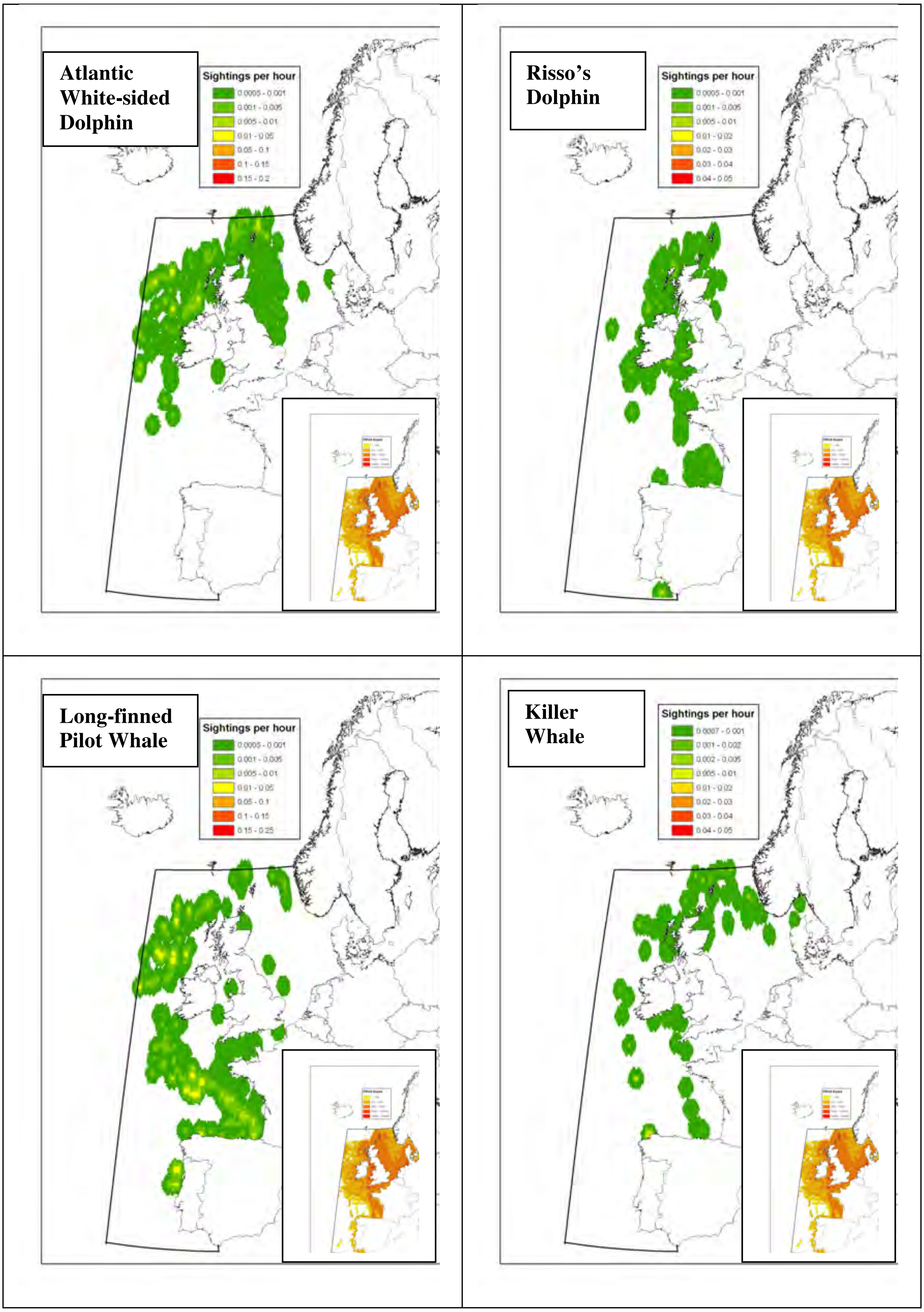


Fig. 6b. Overall Distributions of Cetaceans, 1980-2009
(Inset shows Distribution of Effort)

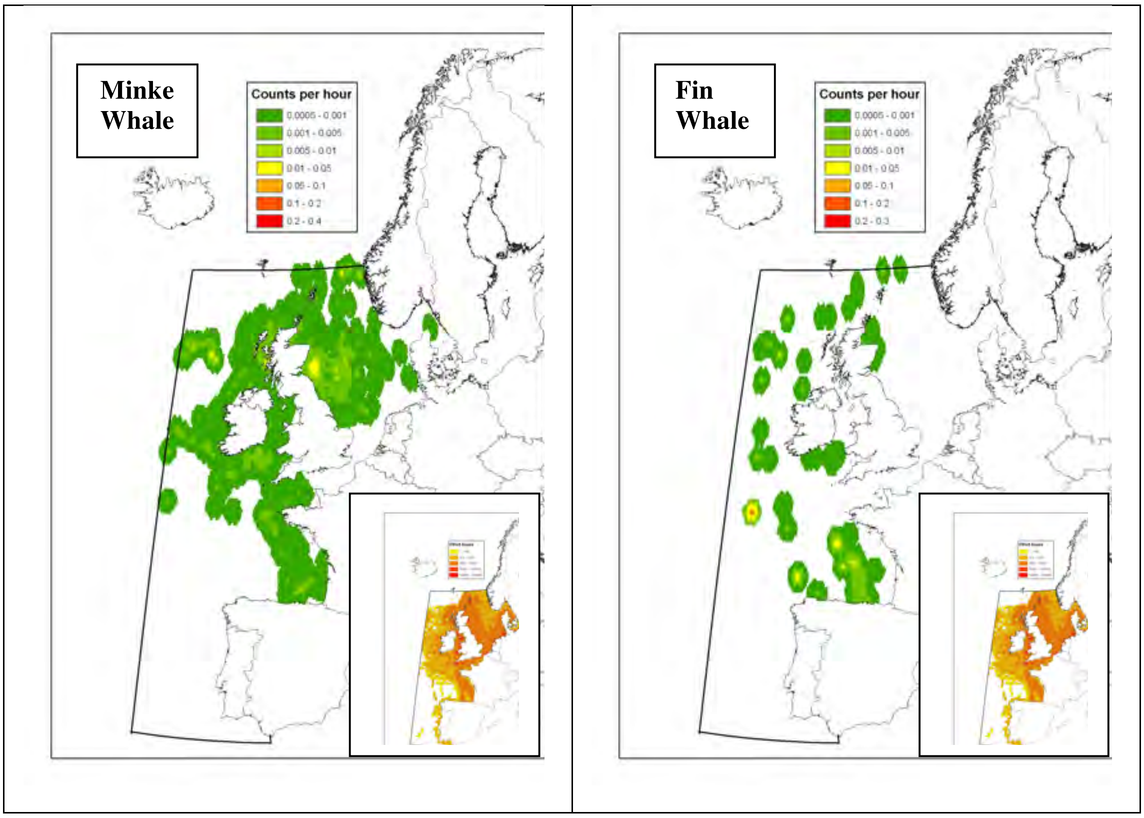


Fig. 6c. Overall Distributions of Cetaceans, 1980-2009
(Inset shows Distribution of Effort)

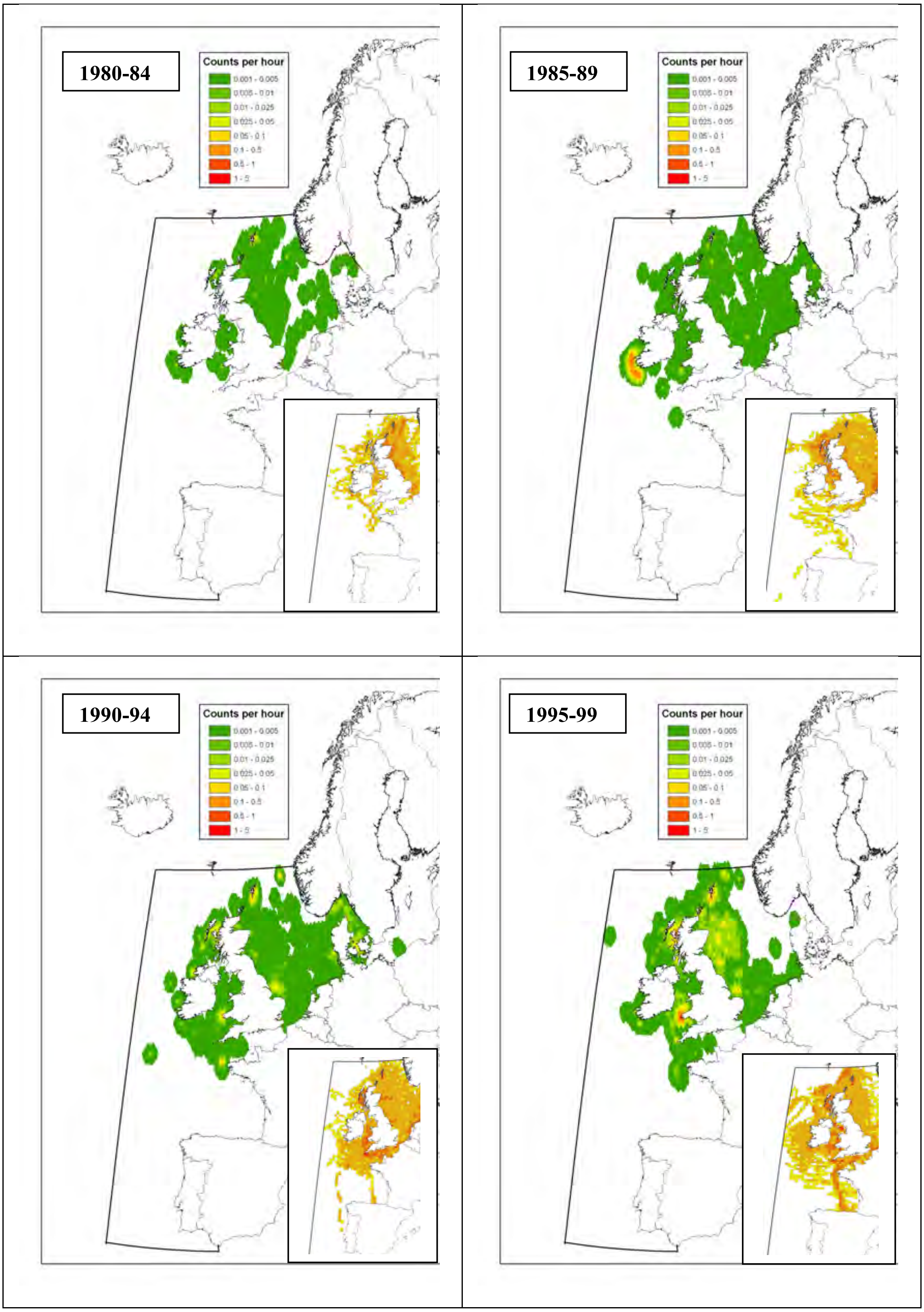


Fig. 7a. Temporal Changes in Distribution of Harbour Porpoises, 1980-99
(Inset shows Distribution of Survey Effort)

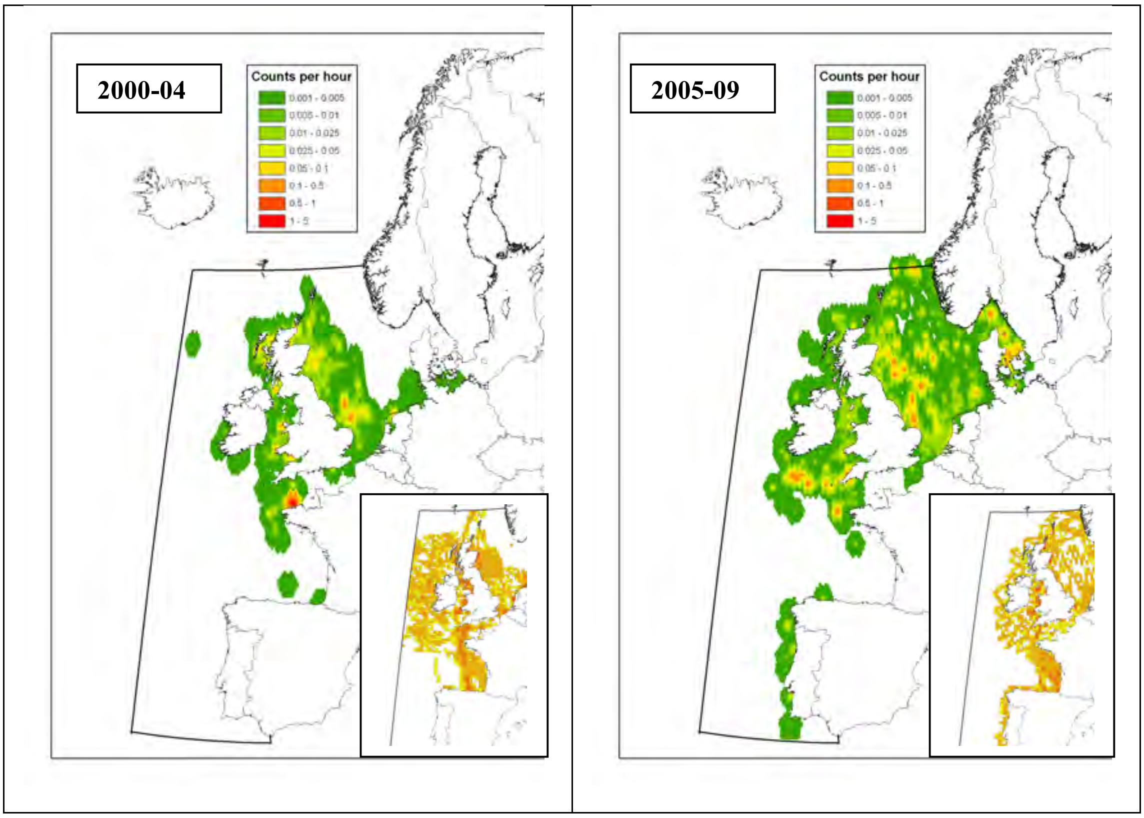


Fig. 7b. Temporal Changes in Distribution of Harbour Porpoises, 2000-09
(Inset shows Distribution of Survey Effort)

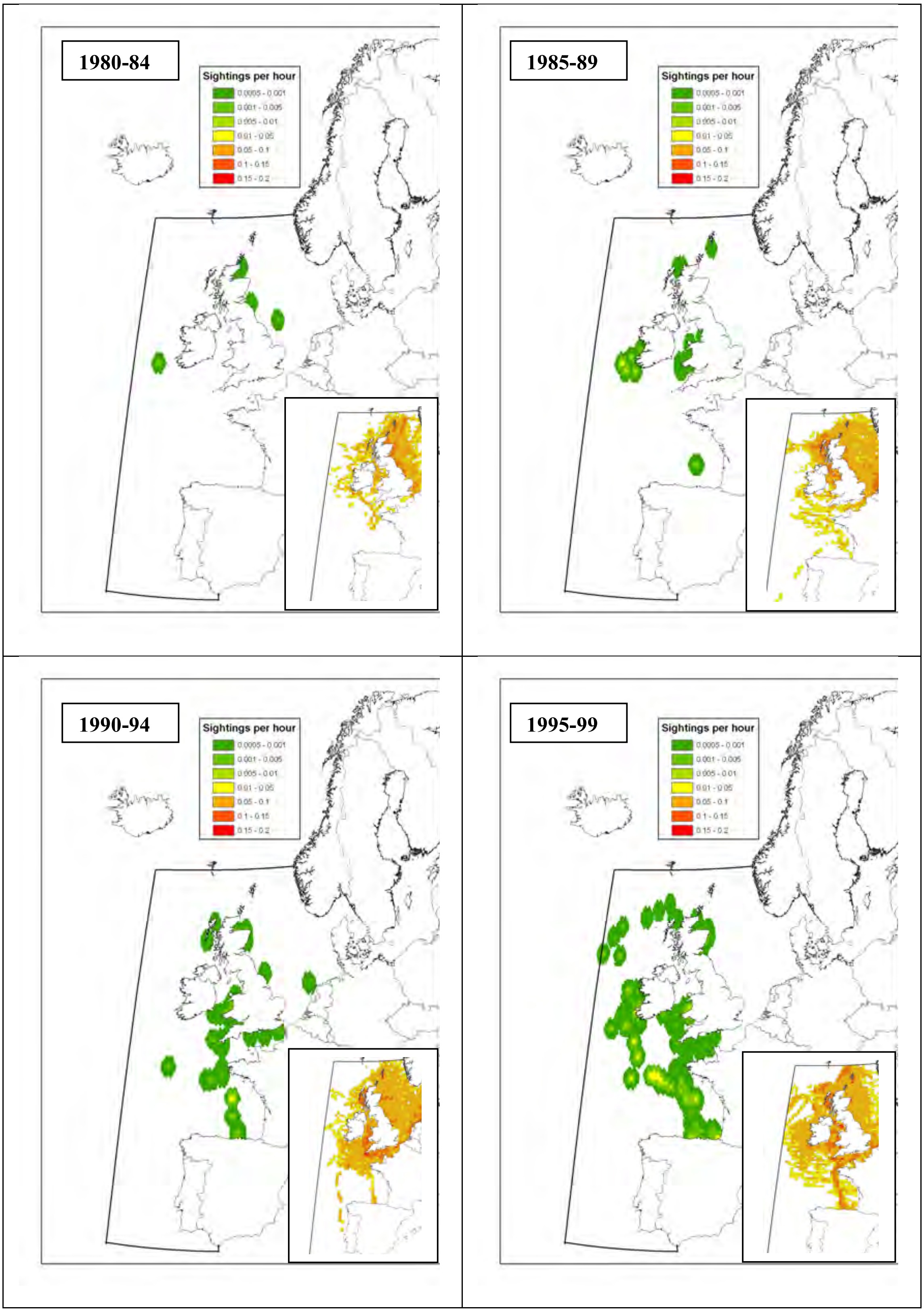


Fig. 8a. Temporal Changes in Distribution of Bottlenose Dolphins, 1980-99
(Inset shows Distribution of Survey Effort)

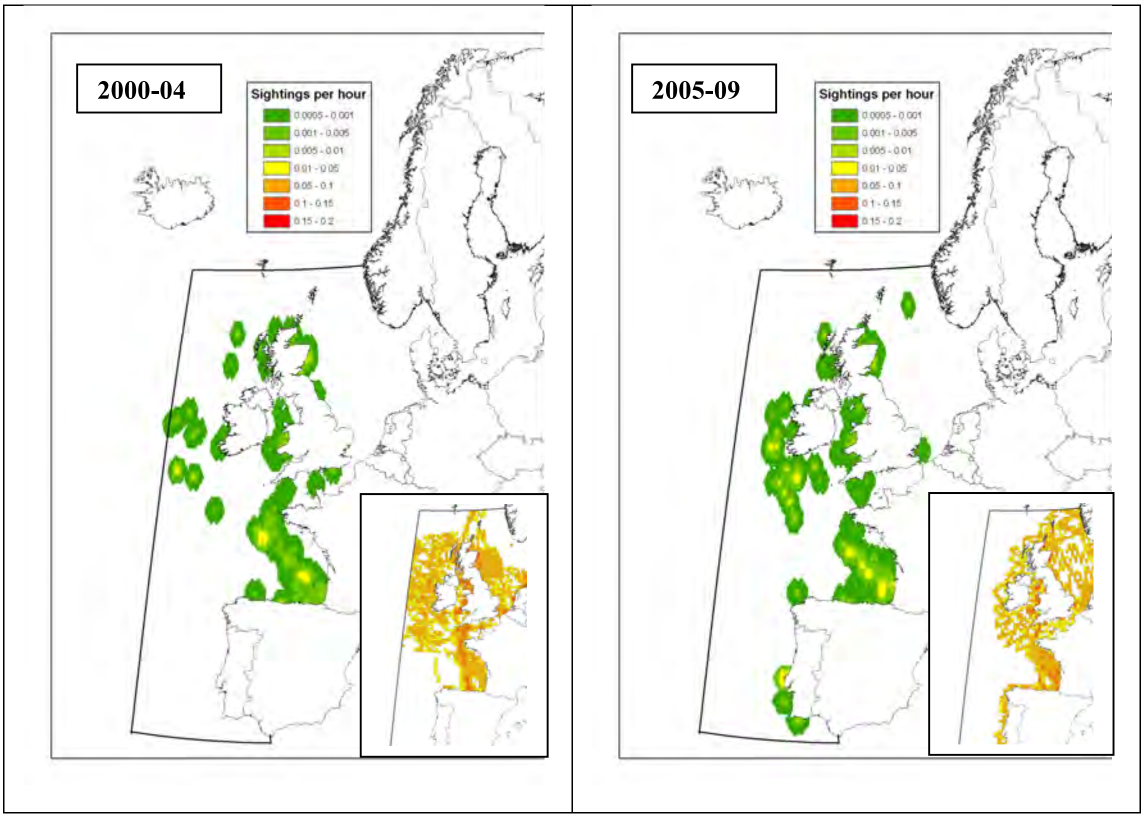


Fig. 8b. Temporal Changes in Distribution of Bottlenose Dolphins, 2000-09
(Inset shows Distribution of Survey Effort)

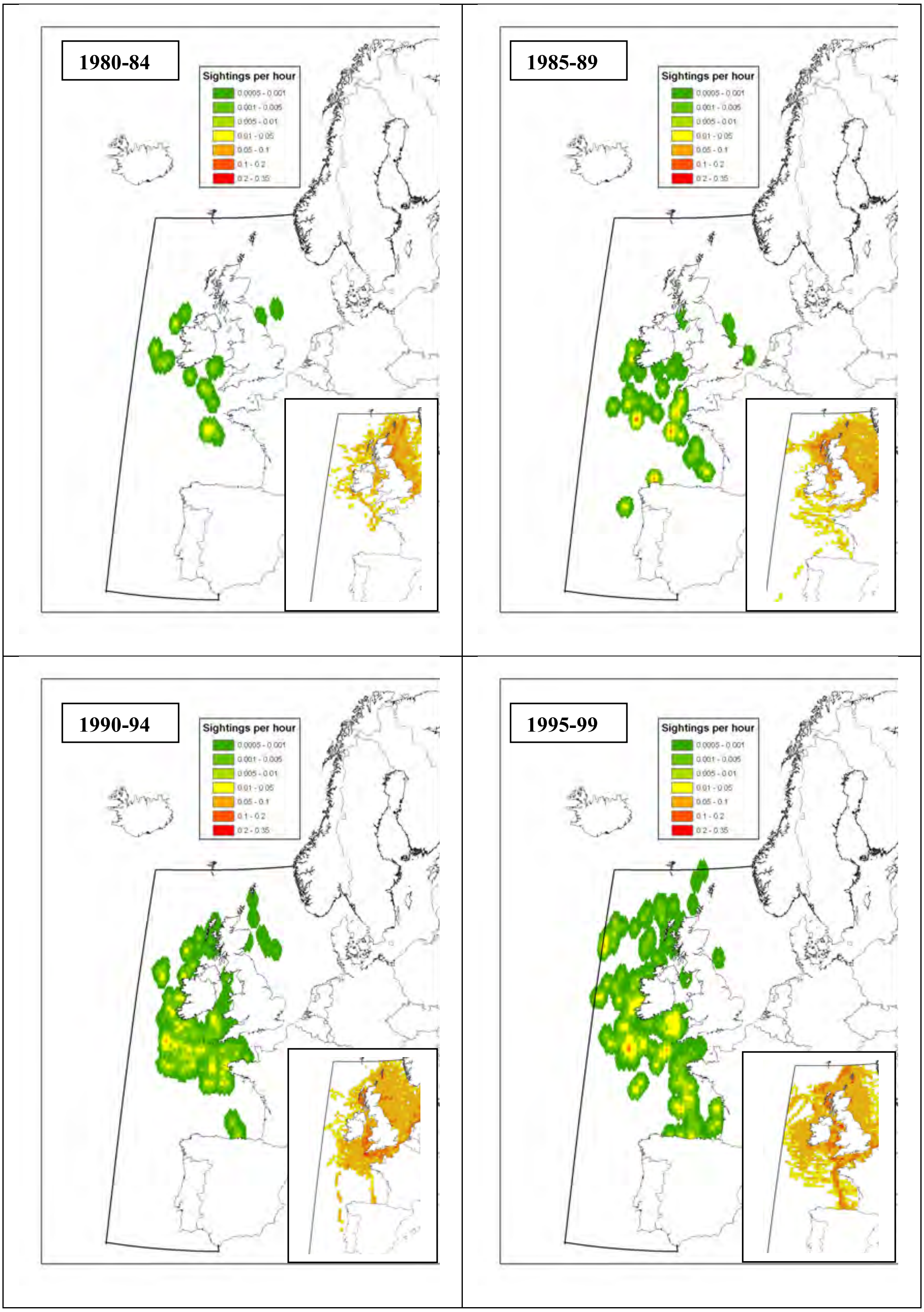


Fig. 9a. Temporal Changes in Distribution of Short-beaked Common Dolphins, 1980-99 (Inset shows Distribution of Survey Effort)

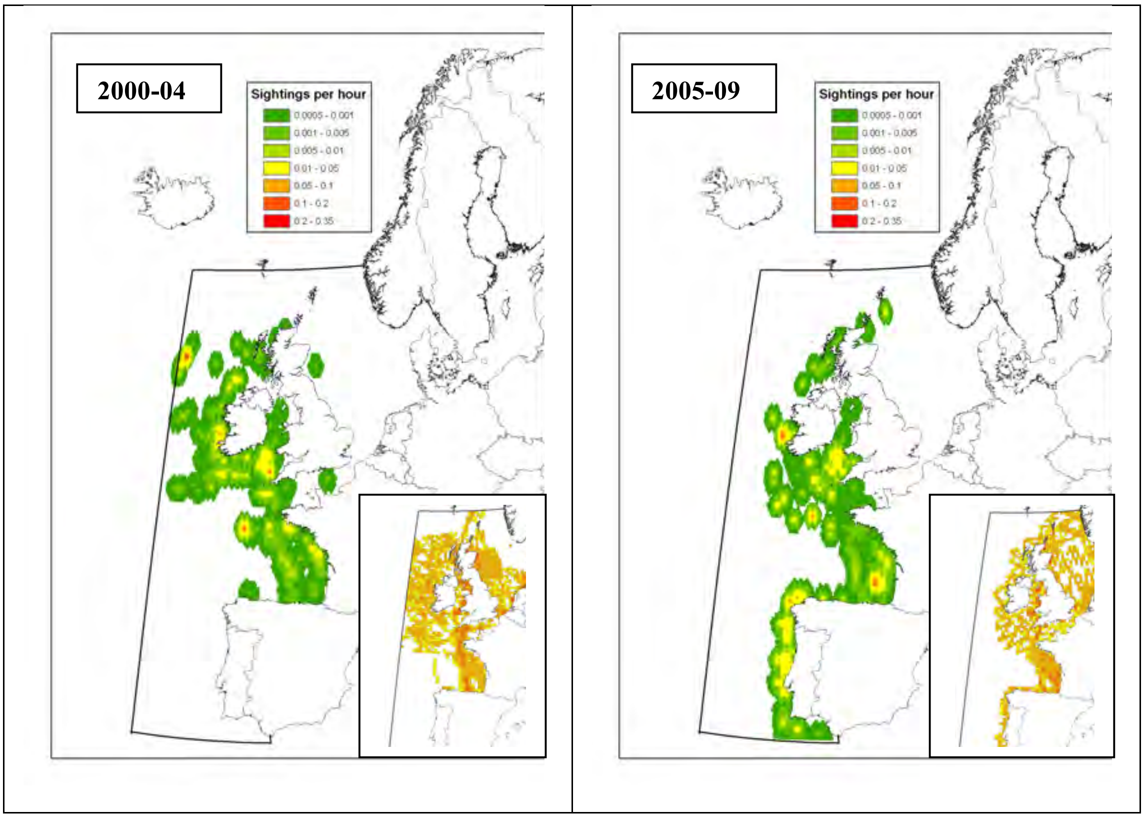


Fig. 9b. Temporal Changes in Distribution of Short-beaked Common Dolphins, 2000-09
(Inset shows Distribution of Survey Effort)

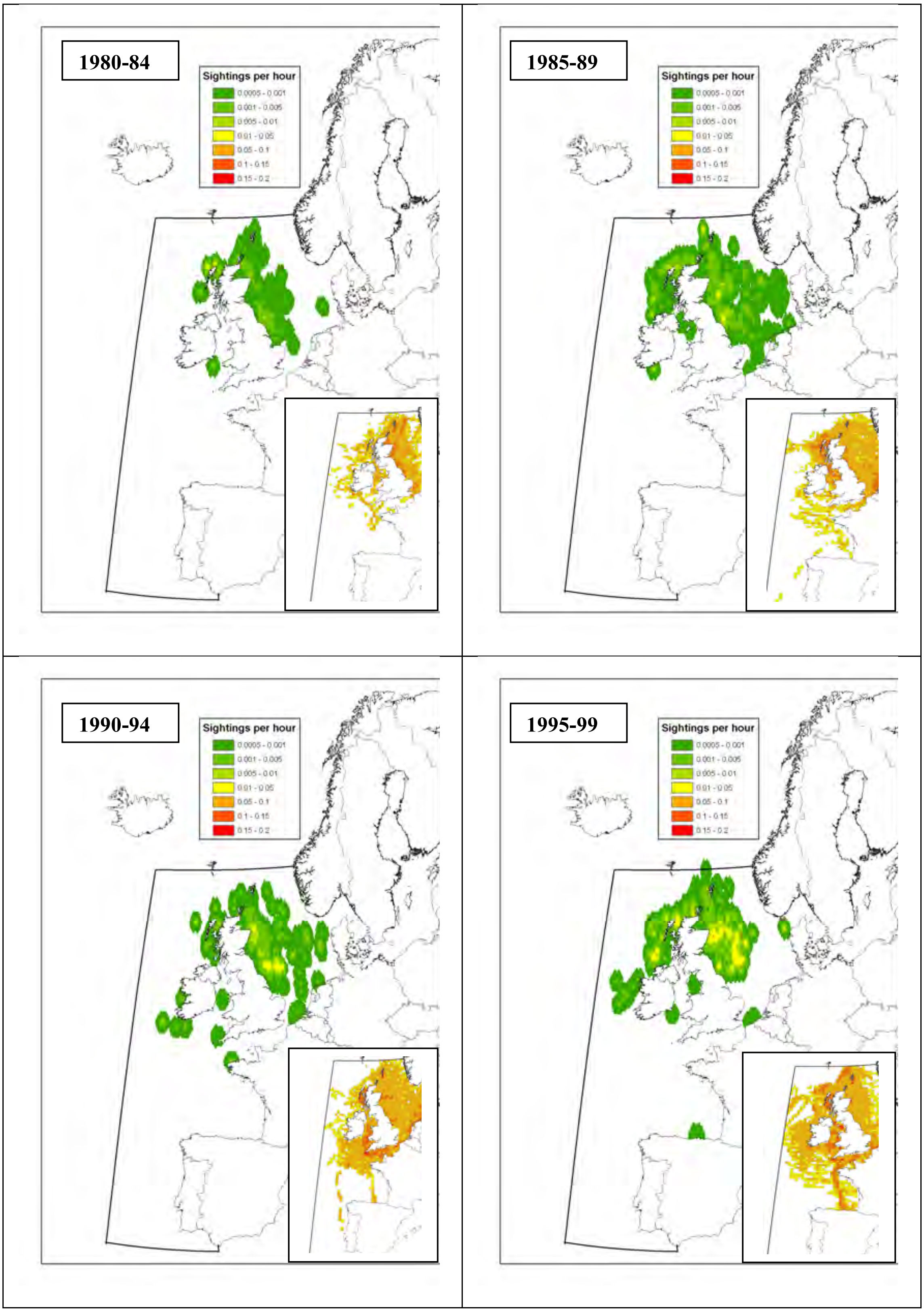


Fig. 10a. Temporal Changes in Distribution of White-beaked Dolphins, 1980-99
(Inset shows Distribution of Survey Effort)

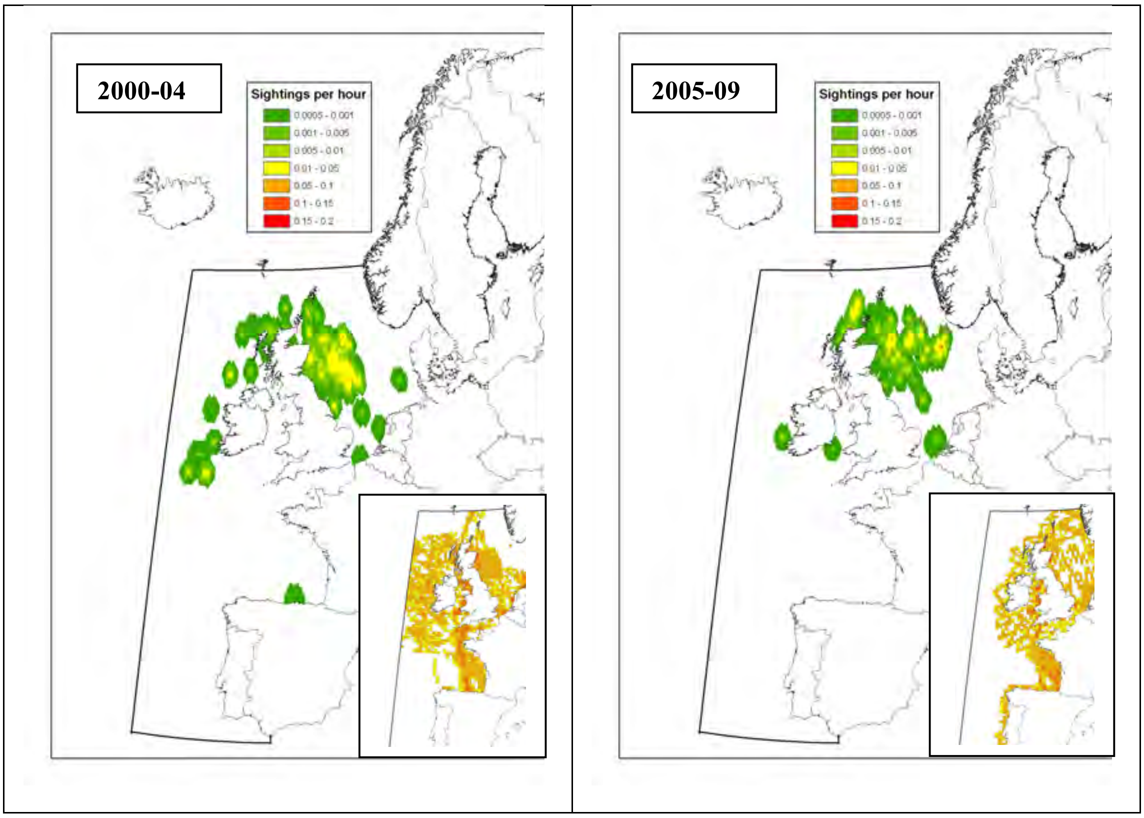


Fig. 10b. Temporal Changes in Distribution of White-beaked Dolphins, 2000-09
(Inset shows Distribution of Survey Effort)

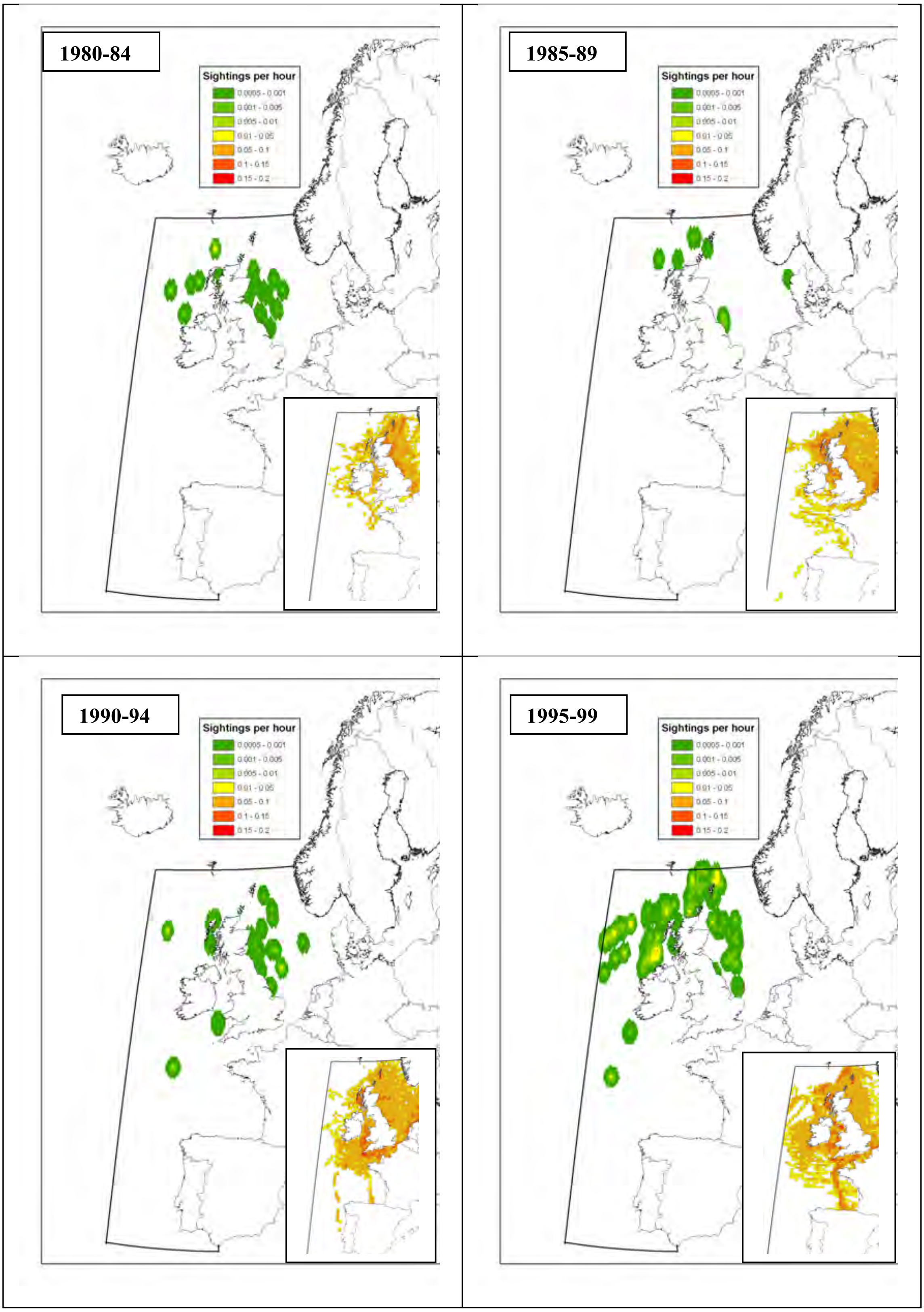


Fig. 11a. Temporal Changes in Distribution of Atlantic White-sided Dolphins, 1980-99 (Inset shows Distribution of Survey Effort)

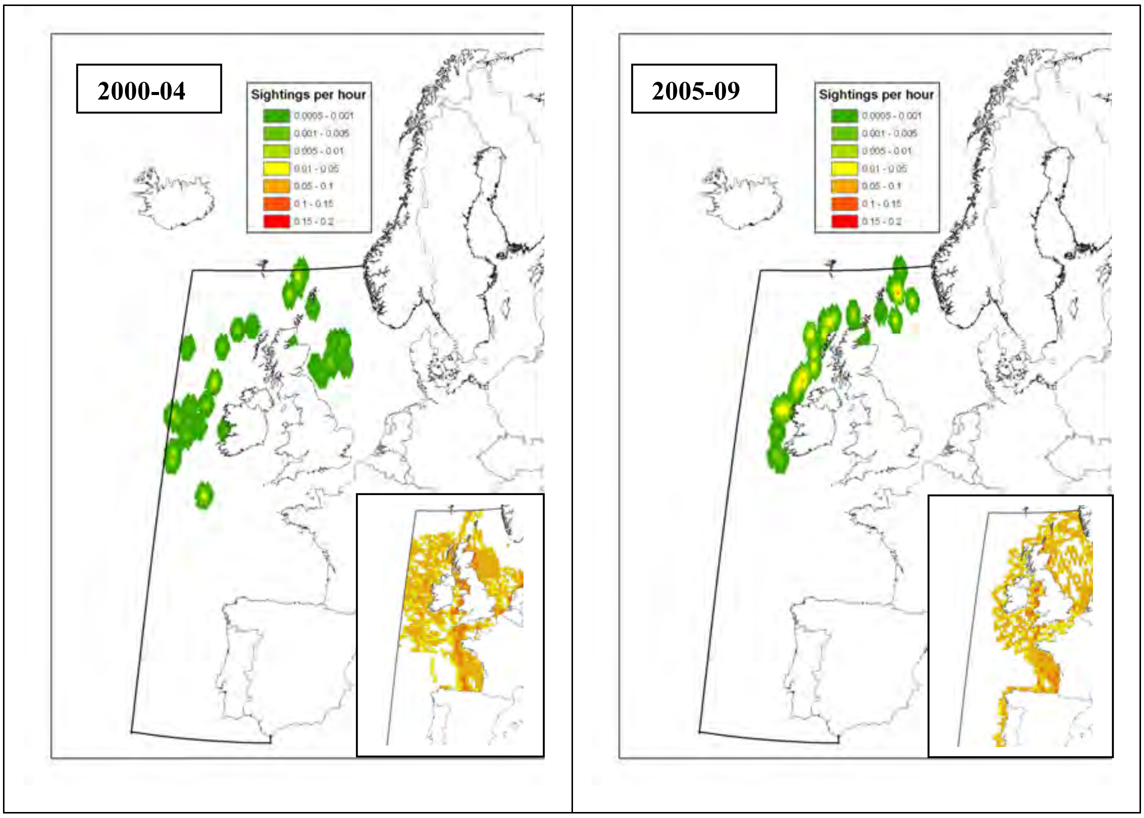


Fig. 11b. Temporal Changes in Distribution of Atlantic White-sided Dolphins, 2000-09
(Inset shows Distribution of Survey Effort)

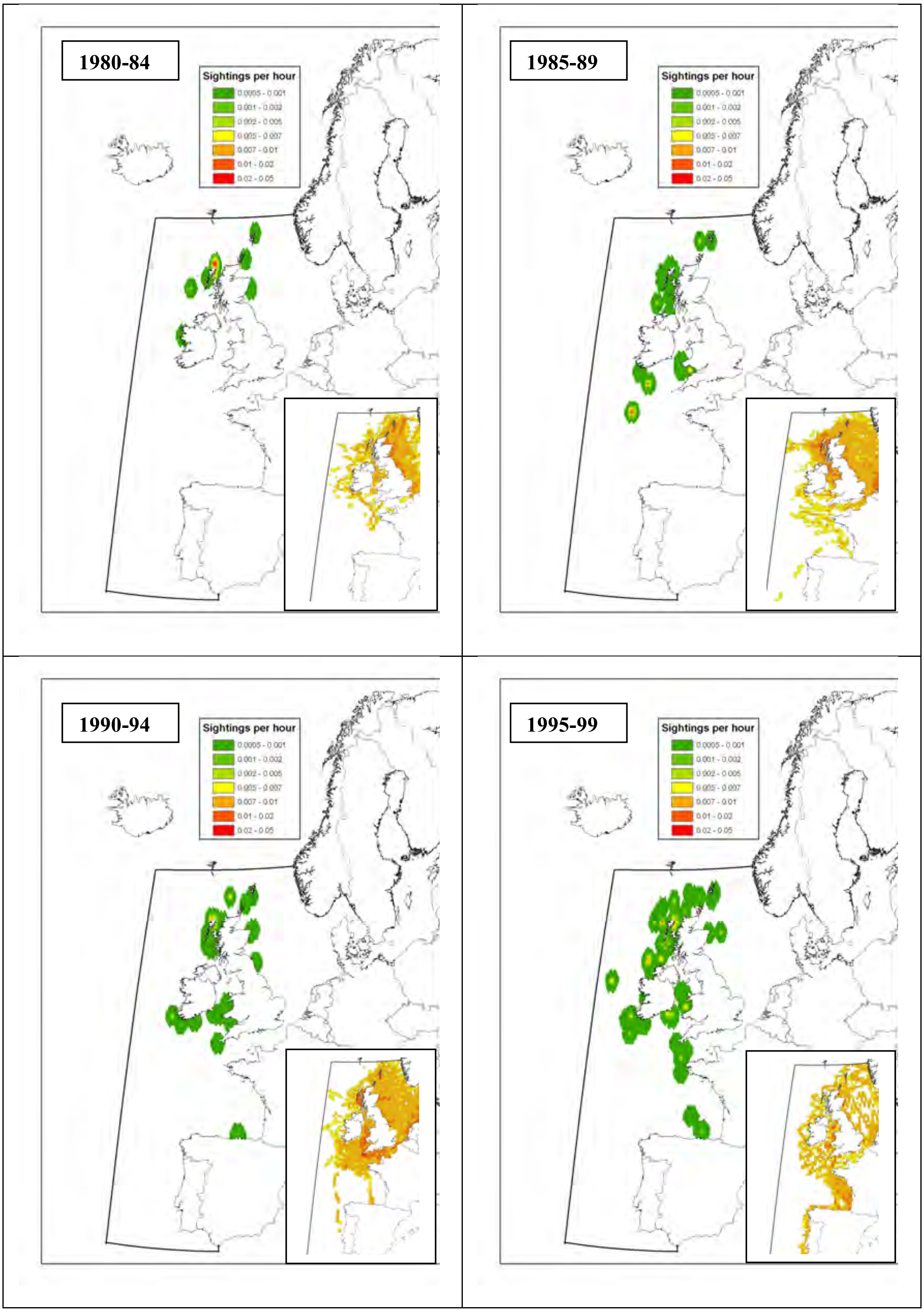


Fig. 12a. Temporal Changes in Distribution of Risso's Dolphins, 1980-99
(Inset shows Distribution of Survey Effort)

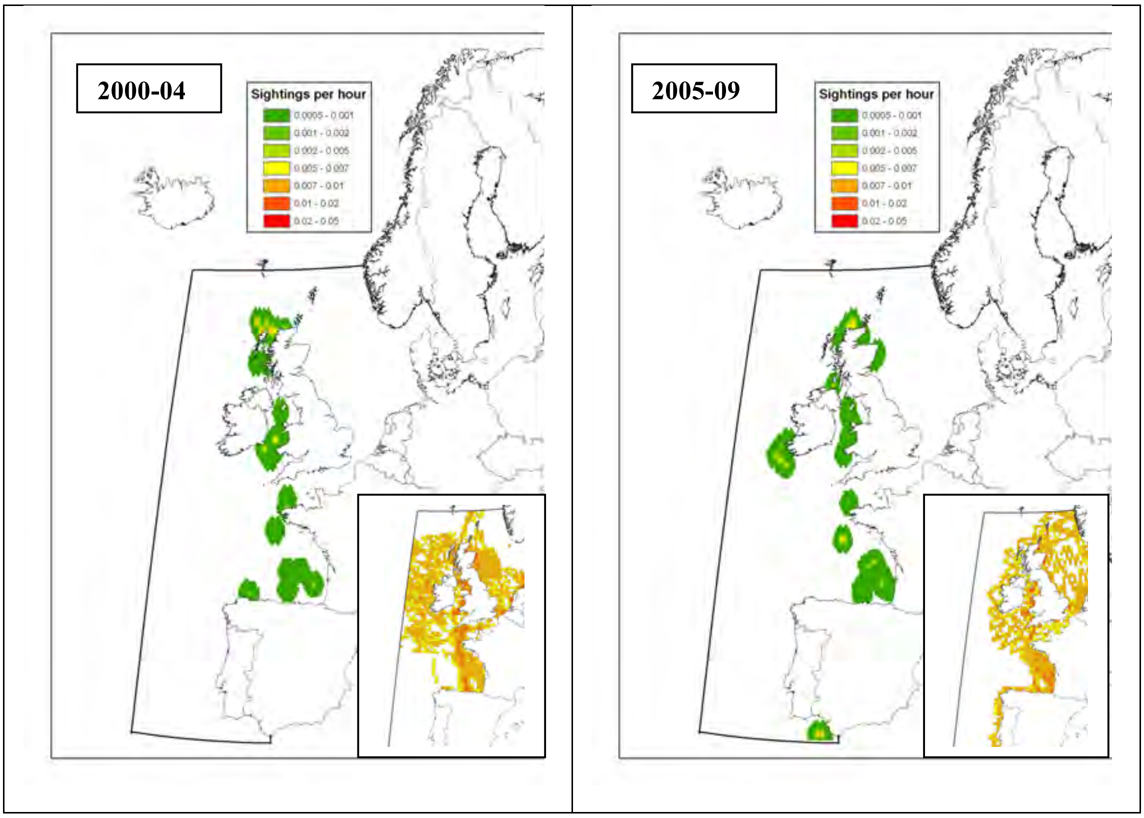


Fig. 12b. Temporal Changes in Distribution of Risso's Dolphins, 2000-09
(Inset shows Distribution of Survey Effort)

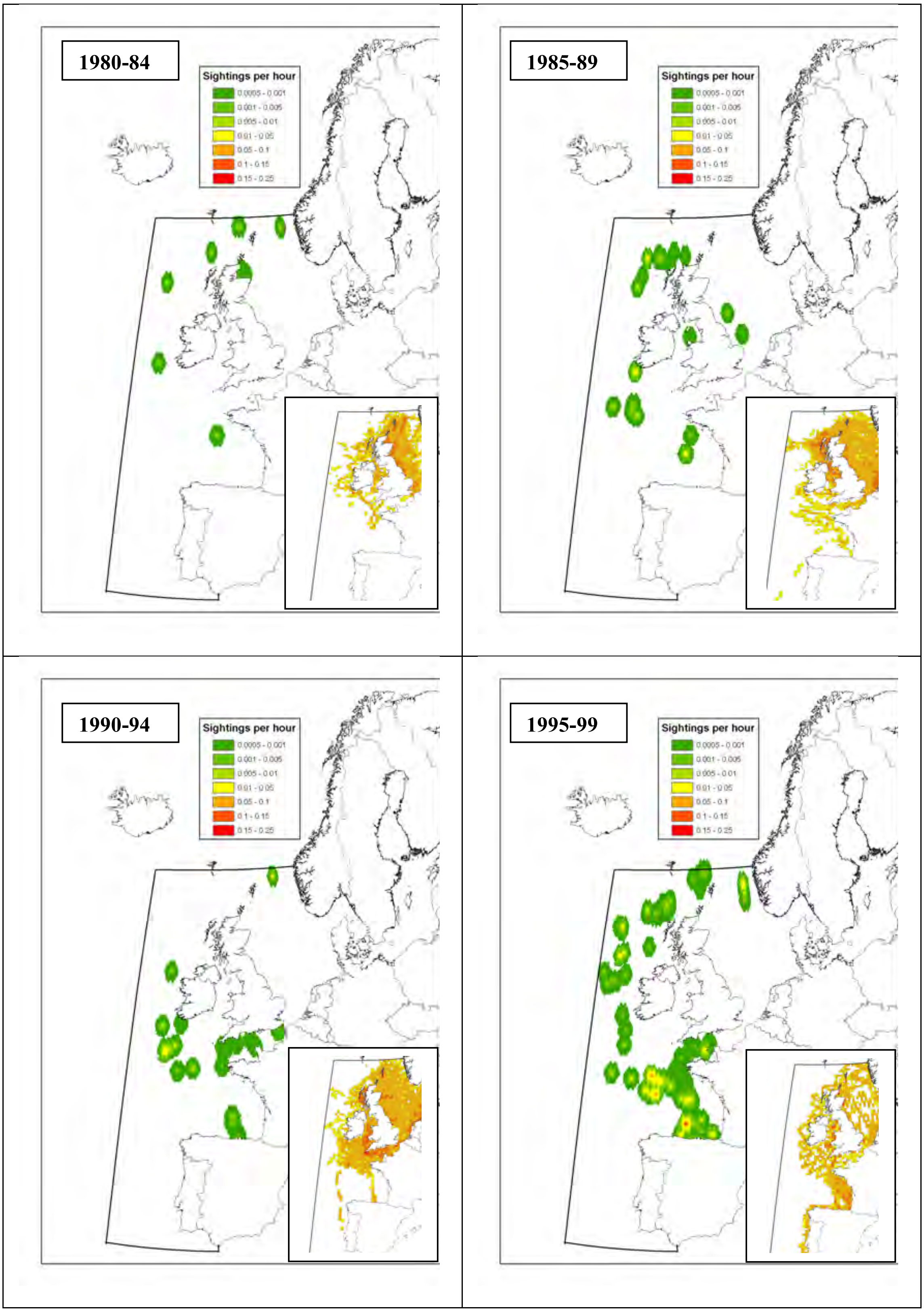


Fig. 13a. Temporal Changes in Distribution of Long-finned Pilot Whales, 1980-99
(Inset shows Distribution of Survey Effort)

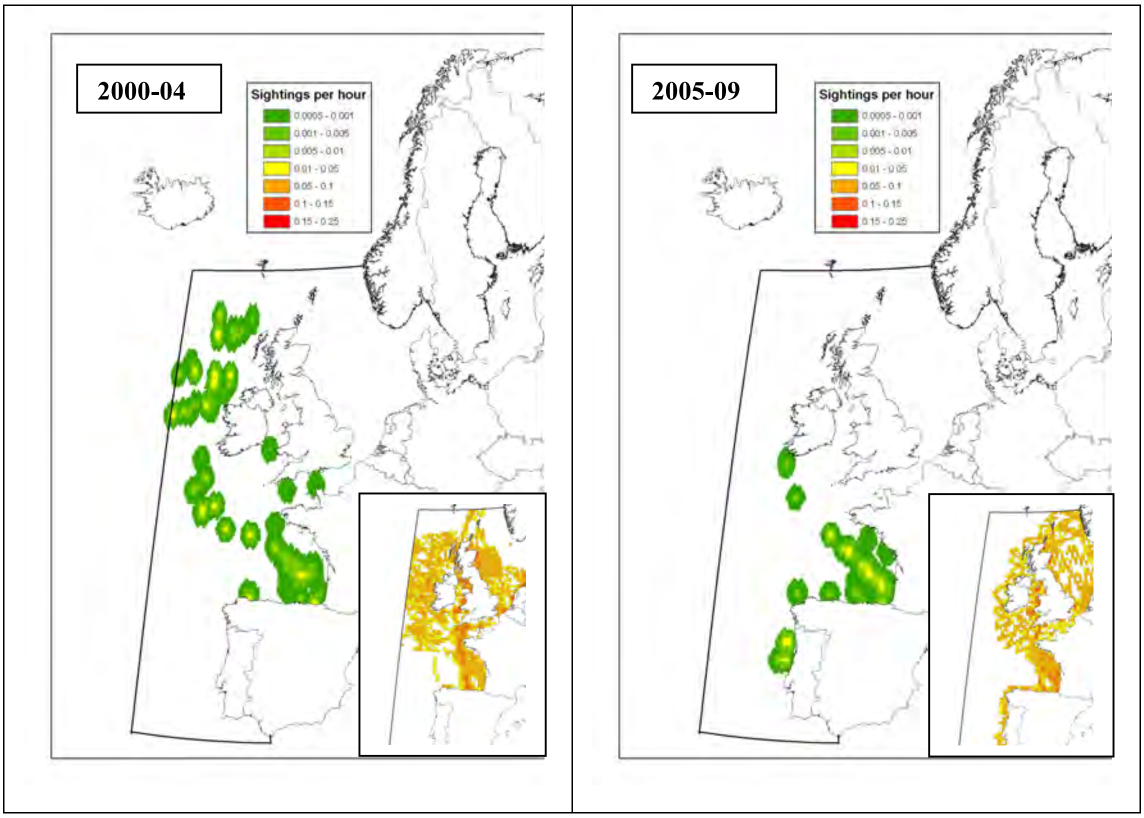


Fig. 13b. Distribution of Long-finned Pilot Whales, 2000-09
(Inset shows Distribution of Survey Effort)

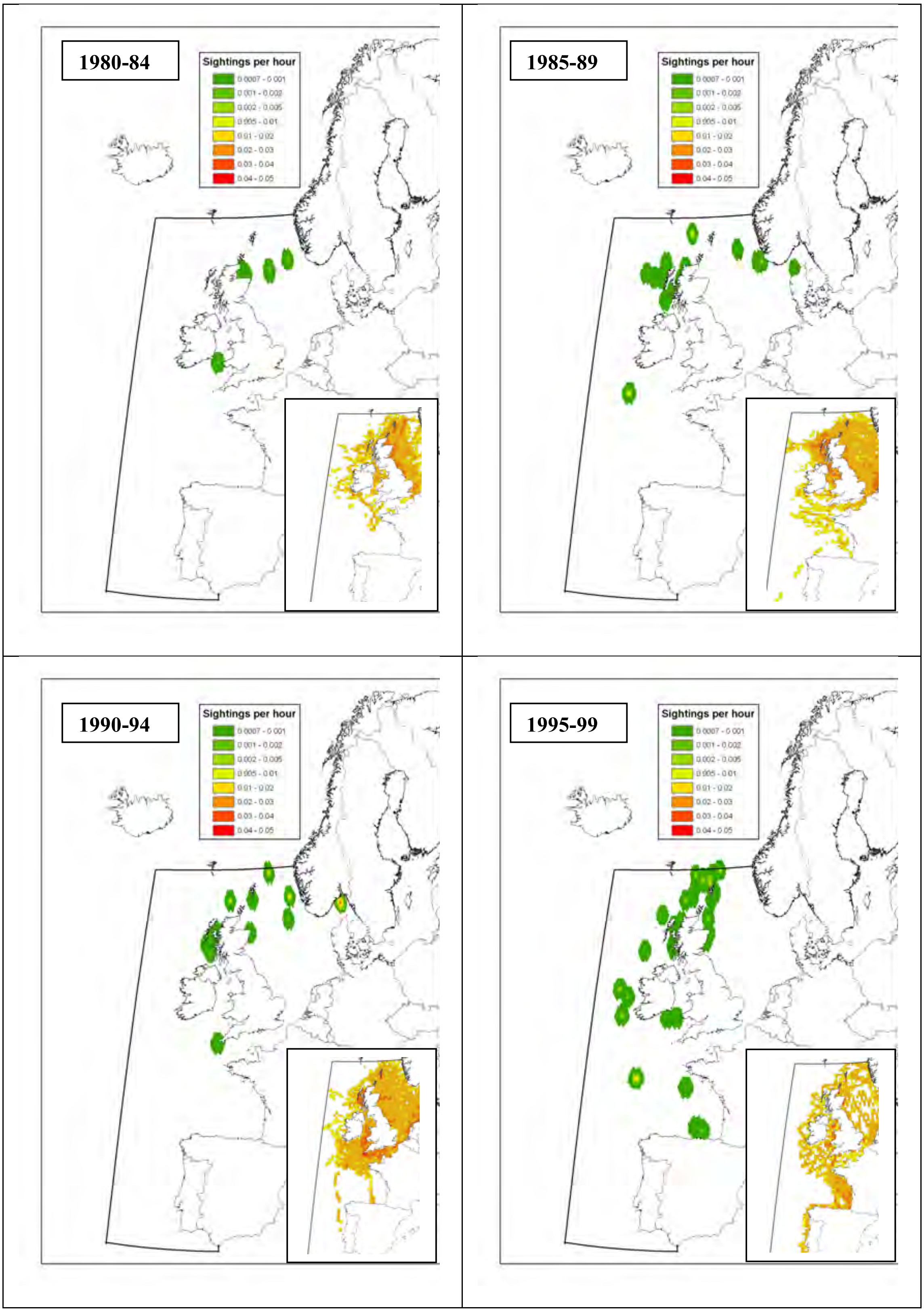


Fig. 14a. Temporal Changes in Distribution of Killer Whales, 1980-99
(Inset shows Distribution of Survey Effort)

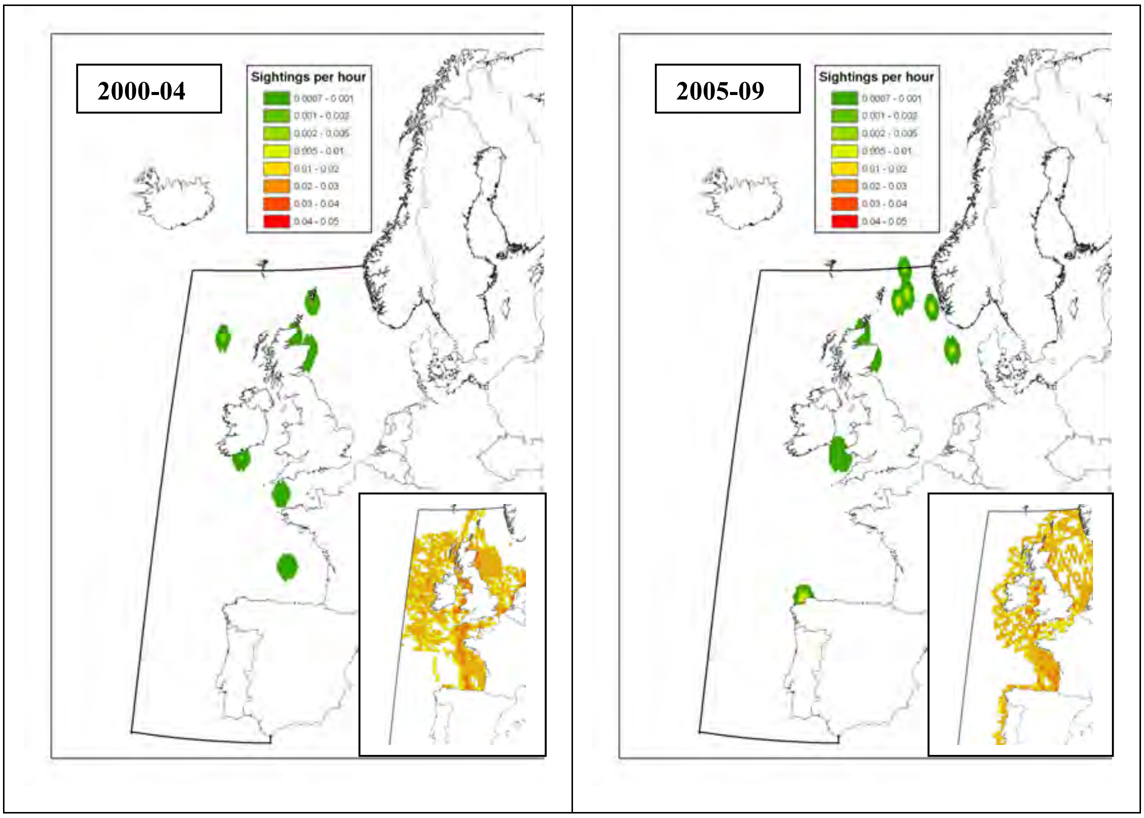


Fig. 14b. Temporal Changes in Distribution of Killer Whales, 2000-09
(Inset shows Distribution of Survey Effort)

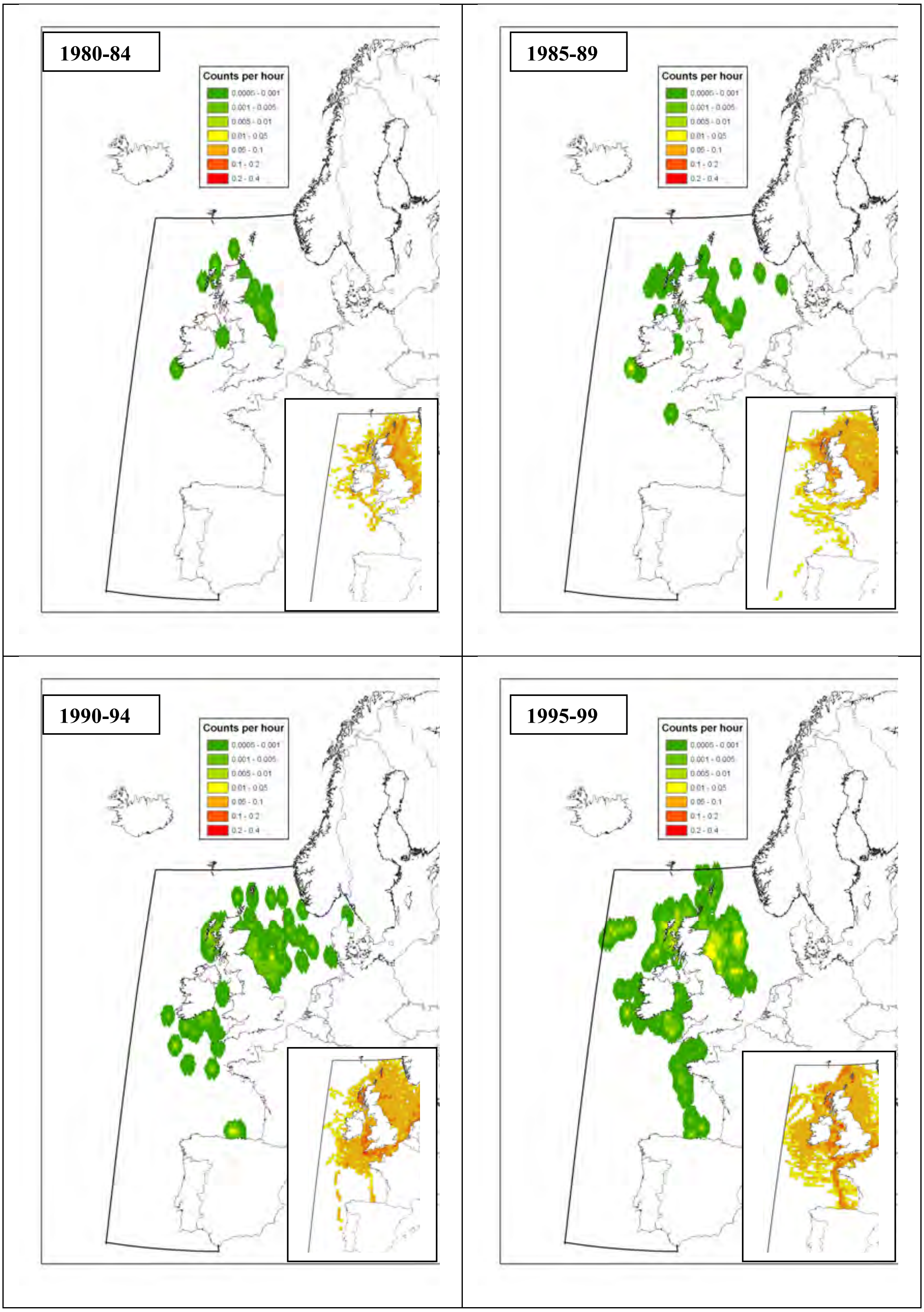


Fig. 15a. Temporal Changes in Distribution of Minke Whales, 1980-99
(Inset shows Distribution of Survey Effort)

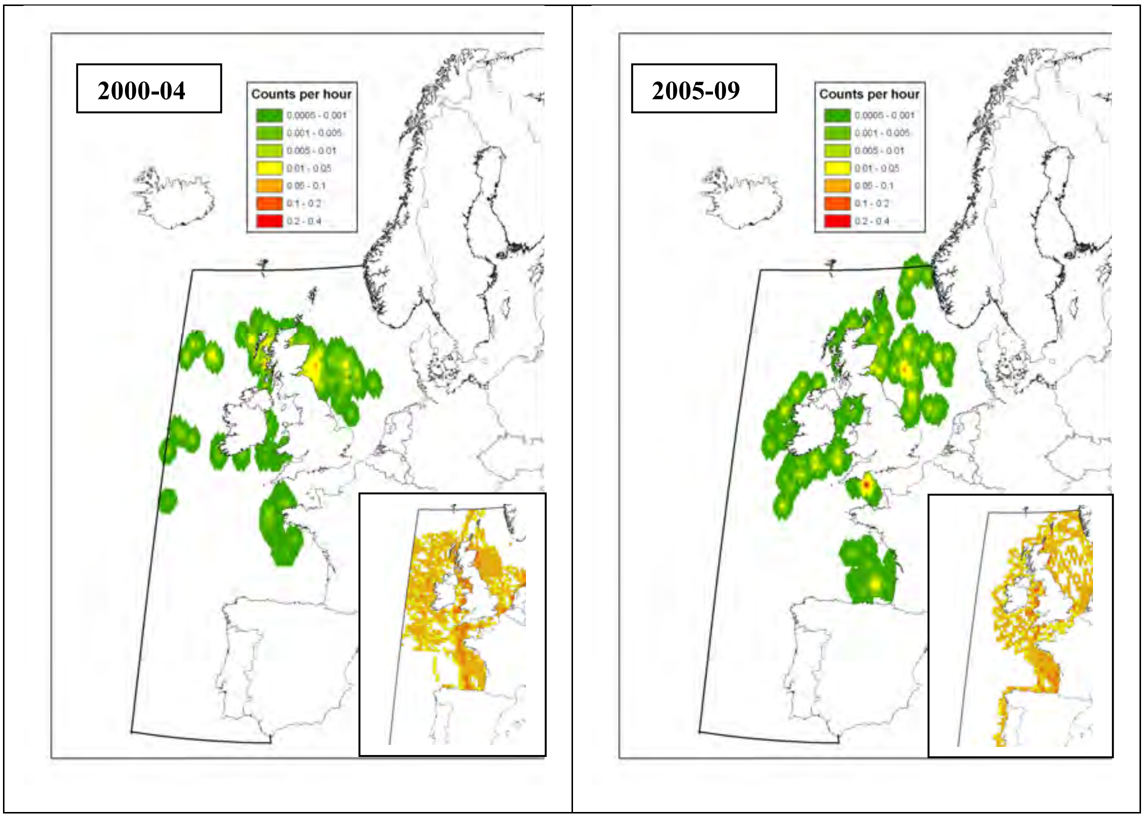


Fig. 15b. Temporal Changes in Distribution of Minke Whales, 2000-09
(Inset shows Distribution of Survey Effort)

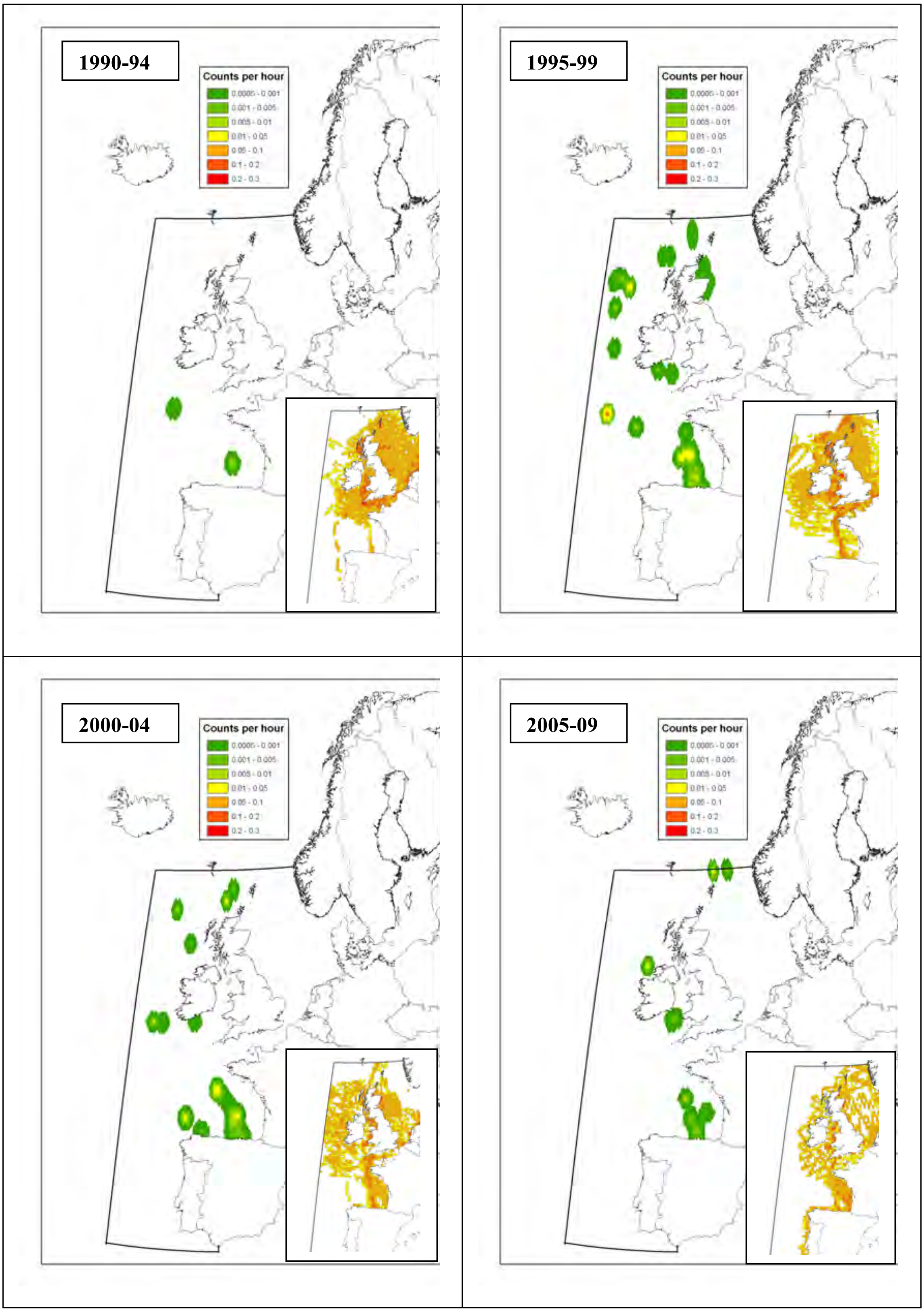


Fig. 16. Temporal Changes in Distribution of Fin Whales, 1990-2009
(Inset shows Distribution of Survey Effort)

3.3 Spatio-temporal trends in Oil and Gas Exploratory Effort

3.3.1 Introduction to Seismic Surveys Marine seismic surveys may be classified as 2D, 3D or 4D types (IOGP, 2011). There may be a number of variants, such as OBC surveys in which cables are laid on the sea bed and a sound source towed over them, but for the purposes of this study the salient feature of interest is the shot point density, or spatial density of ensonification by the seismic source, for which the 2D / 3D / 4D classification is sufficient.

The primary aim of a 2D survey is to create two-dimensional sections through the rock layers underlying the seabed, each survey line taking a linear geological slice. Survey lines tend to be relatively long and may be transected by further 2D lines, often perpendicular to the main line. Shot point intervals are regularly spaced, typically at 25m and the vessel moves slowly, such that the interval between shots is usually a little less than 10 seconds. The resulting ensonification may therefore extend over a large area, but at any one point in the survey area activity tends to be transient, unless the survey vessel returns to acquire a line through that point on a different azimuth.

The aim of a 3D survey is to compile a three-dimensional image of the geological formations under a study area. To achieve this, closely spaced parallel transect lines are surveyed in order to acquire full spatial coverage of the study area. The shot point density of a 3D survey is therefore significantly higher than for a 2D survey and activity tends to last much longer within the survey area. Source sizes are variable but tend to be similar between 2D and 3D survey types.

A 4D survey is simply an exact repetition of an earlier 3D survey, so for the purposes of this study we make no distinction between 3D and 4D surveys.

For further details of marine seismic survey operations, see IOGP (2011).

3.3.2 Spatio-temporal trends The first 2D survey was carried out in 1959 in Dutch waters of the North Sea. Annual numbers of surveys increased to a peak of nearly 200 in 1972, before falling to less than half that number in the late 1970s. 2D exploration picked up again through the 1980s and early 1990s, but then became gradually less frequent. The first 3D survey was carried out in 1977, but this did not become the dominant type of survey until the 1990s, reaching a peak in the middle of that decade.

Shotpoint densities were low throughout the early 1960s, and mainly in the southern North Sea (Fig. 17a). Seismic activity extended northwards in the North Sea during the late 1960s and early 1970s. During the 1970s, seismic surveys also took place throughout the Irish Sea and around Ireland, with coastal activities additionally around the Iberian Peninsula (Fig. 17a). The same areas received seismic activity through the 1980s and 1990s, with peak shotpoint densities in the North Sea between 1985 and 1994 (Fig. 17b). From 1995 onwards, seismic surveys here steadily declined, whilst starting up along the Atlantic Frontier on the edge of the continental shelf west of Britain and Ireland. However from 2000 onwards, there was relatively little seismic survey activity anywhere in the ASCOBANS region, with that effort mostly concentrated in the same central area of the North Sea as in earlier periods Fig. 17c). This is highlighted when combining data for all time periods, particularly with respect to 2-D surveys (Fig. 17c).

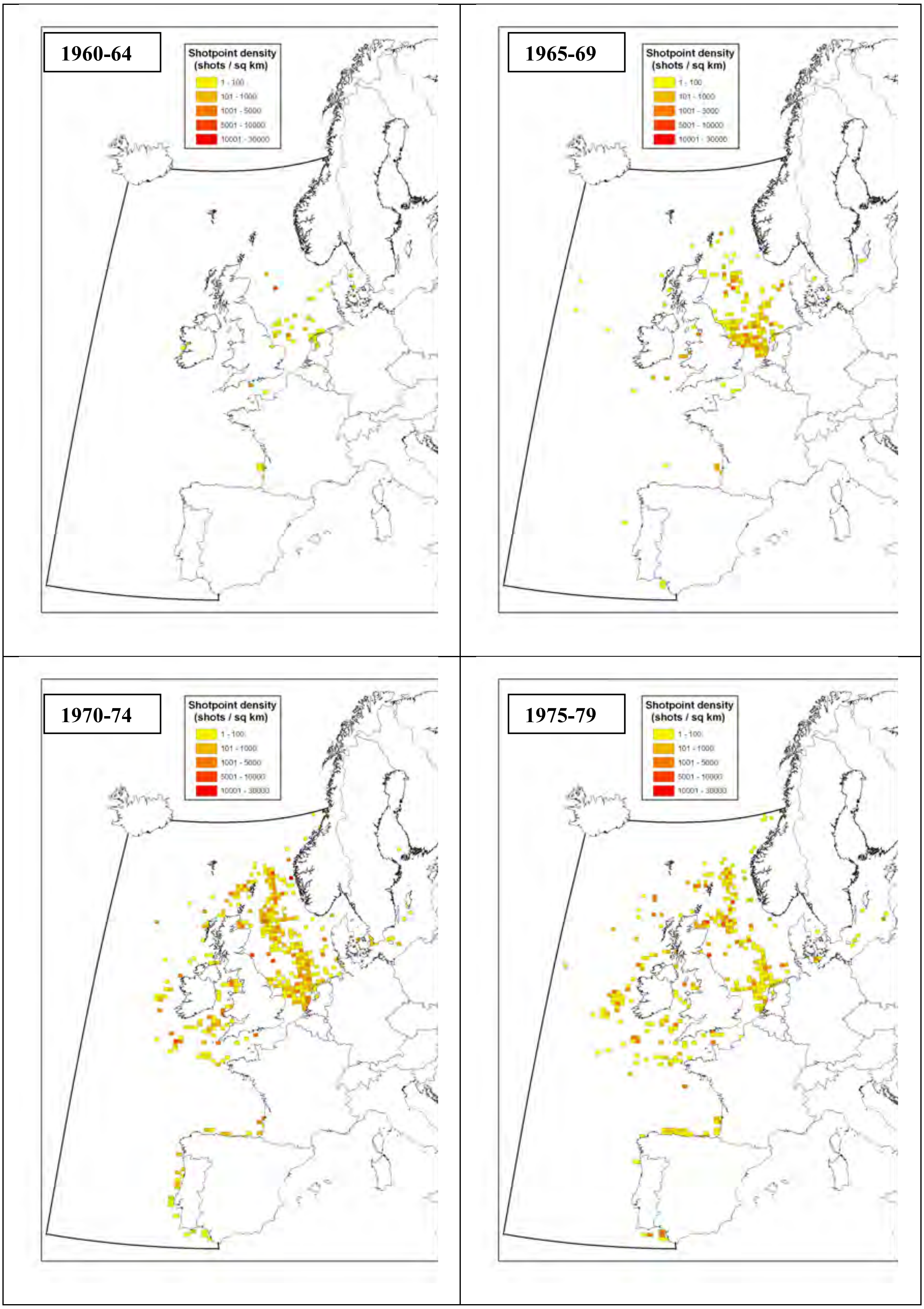


Fig. 17a. Seismic Shotpoint Densities, 1960-79

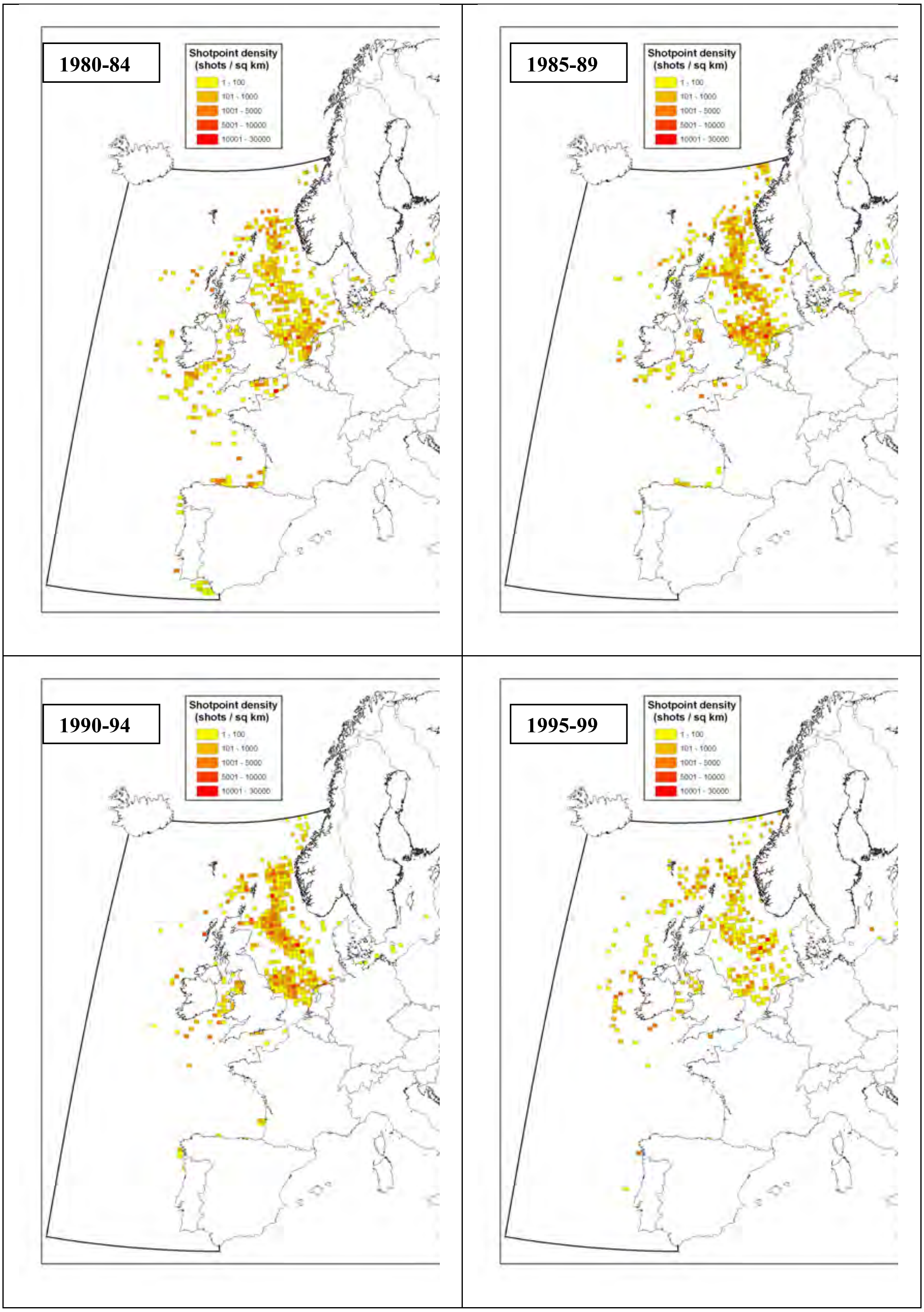


Fig. 17b. Seismic Shotpoint Densities, 1980-99

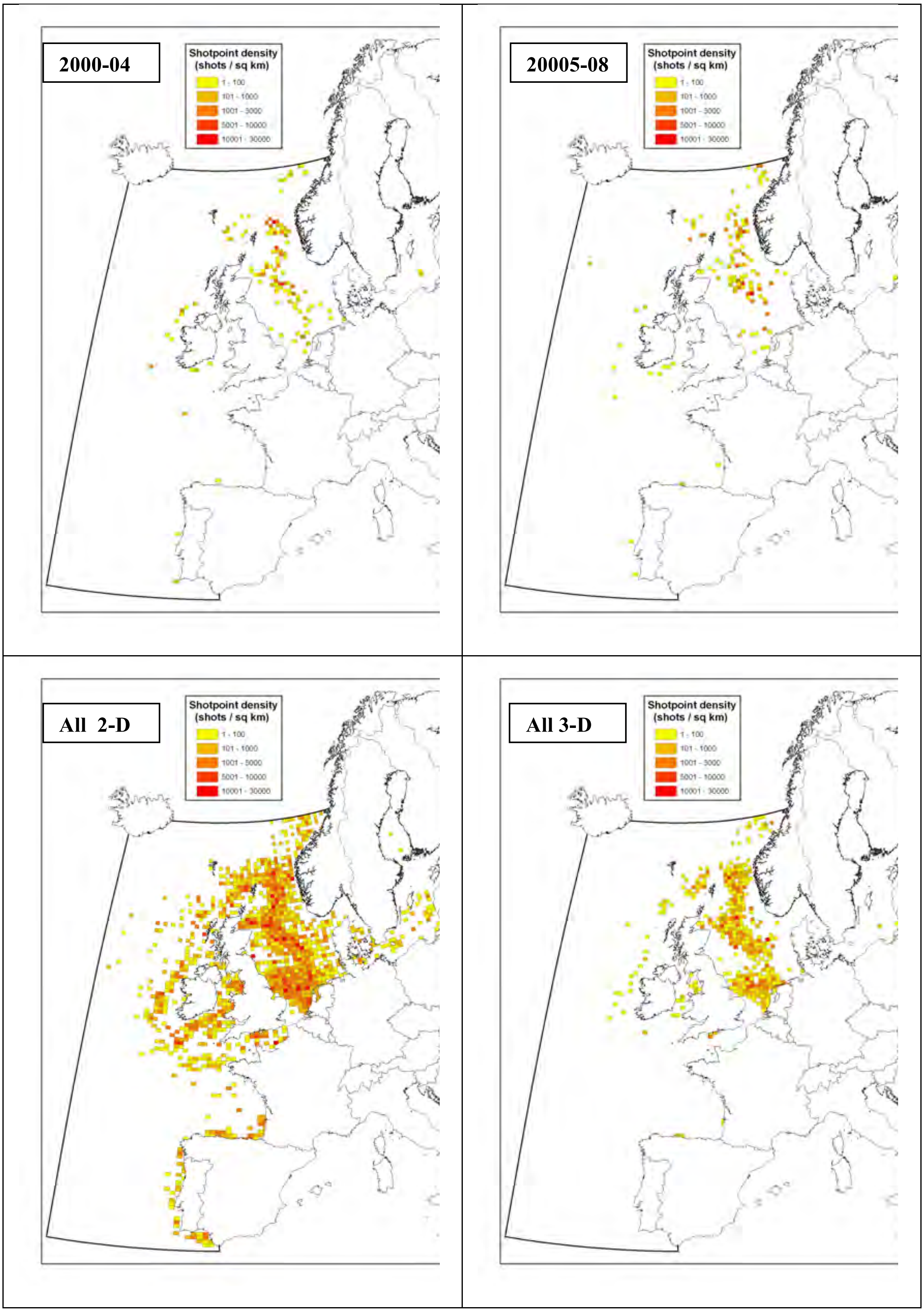


Fig. 17c. Seismic Shotpoint Densities, 2000-08, and all 2-D and 3-D Surveys

3.4 Cetacean Trends in Relation to Oil and Gas Exploration

3.4.1 Methods

We built two models for each species. All models were run as GAM's with a binomial distribution (i.e. presence – absence models), using a forward selection based on the UBRE score. Explanatory variables were included in the following order: Shot density, Effort, Time period and Lat*Long (i.e. the interaction between Lat and Long to account for spatial autocorrelation). For the first model (a), we used all data from cells with observer effort, i.e. also including sighting rates of 0. For the second model (b), we only included cells with seismic survey activity during at least one time period. The baseline was the time period 1980-84, and trends for each subsequent time period were then compared with that.

3.4.2 Results

No species has experienced a decline in sighting rates over time (see Fig. 18). Although the chance of detecting each species was positively correlated with observer effort in both models (except for fin whales in model b), the significant effects of lat*long in all and time period in most (16 out of 20) models indicates that effort alone was not sufficient in explaining presence/absence per cell, and the observed temporal trends.

Model a) Shot density was significant only for white-beaked (positive correlation) and common dolphin (negative correlation), when it was included in the model together with effort and time period. However, when lat*long was added to the model, shot density became non-significant and was excluded in the case of common dolphin. The presence/absence of all species was thus determined mainly by the position of the cells, the amount of effort and time period, while shot density itself was never significant when all parameters were included. In some models, it was retained for better fit, in others excluded completely. As the combination of location and time period already explains most of the variation in the data that would be included in shot density, this result was not surprising. However, the forward selection with lat*long included last also enabled an examination of shot density alone first.

Model b) The results were similar to model a): Shot density never reached significance in the GAMs for presence – absence except for harbour porpoise when only shot density and effort were included in the model ($p=0.0018$, with a negative correlation). This significance became stronger when lat*long was added, but disappeared as soon as time period was included; the latter was the best model according to the UBRE score.

In both models, significant positive increases from the 1980-84 baseline were observed for the majority of time periods in minke whale, common dolphin, white-beaked dolphin, Risso's dolphin, bottlenose dolphin, and harbour porpoise.

Table 5a. GAM results for all cells with survey effort (n= 8365; including sighting rates of 0). X^2 values are listed for continuous variables, Z values for the categorical variable time period. ***: p<0.001, **: p<0.01, *: p<0.05. n.s. = included for better fit of the model, but not significant; - = not included in final model.

					Species					
Parameters	**FW**	**MW**	**KW**	**LFPW**	**RD**	**AWSD**	**WBD**	**SBCD**	**BND**	**HP**
Shot density	-	6.20	-	-	-	-	0.84	-	-	0.71
Effort	103.50***	145.06***	154.20***	66.09***	113.30***	110.00***	74.14***	148.20***	128.80***	183.13***
LatxLong	396.50***	440.22***	113.50***	126.96***	150.00***	236.20***	314.23***	394.80***	172.00***	312.63***
Time period	n.s.			n.s.						
1985-89	0.00	1.61	0.81	-0.75	-1.70	-3.40***	2.99**	-0.67	0.94	-0.96
1990-94	0.00	5.94***	0.96	-0.85	-1.19	0.91	2.375*	4.76***	2.38*	2.83**
1995-99	0.00	9.03***	3.76***	0.62	2.92**	6.55***	4.09***	4.54***	4.10***	5.34***
2000-04	0.00	7.63***	1.00	0.10	1.50	1.18	2.78**	3.01**	3.96***	5.42***
2005-09	0.00	8.58***	2.45*	-0.18	2.72**	4.86***	0.51	4.17***	4.81***	9.64***
Deviance explained	50.4%	29.6%	22.0%	31.7%	32.0%	27.4%	36.1%	33.6%	35.0%	31.3%

Table 5b. GAM results for cells with seismic shots only (n = 3015). X^2 values are listed for continuous variables, Z values for the categorical variable time period. ***: p<0.001, **: p<0.01, *: p<0.05. n.s. = included for better fit of the model, but not significant; - = not included in final model.

					Species					
Parameters	**FW**	**MW**	**KW**	**LFPW**	**RD**	**AWSD**	**WBD**	**SBCD**	**BND**	**HP**
Shot density	-	-	-	-	-	-	-	-	0.98	-
Effort	-	113.90***	107.28***	29.76***	102.40***	47.89***	88.17***	44.48***	47.34***	193.40***
LatxLong	13.35**	378.30***	77.53***	105.48***	151.40***	125.67***	343.69***	269.60***	100.38***	310.60***
Time period	-									
1985-89		0.64	-1.95	-1.07	2.89**	-0.03	3.10**	0.08	-1.58	-0.66
1990-94		5.43***	0.13	-1.47	3.45***	1.28	2.92**	4.39***	1.21	2.99**
1995-99		8.72***	1.33	0.99	4.45***	5.14***	4.30***	4.43***	2.09*	7.95***
2000-04		7.57***	0.88	-0.44	3.44***	0.97	4.00***	2.19*	2.45*	7.53***
2005-09		9.57***	3.83***	0.00	3.60***	2.42*	2.38*	2.69**	3.51***	12.86***
Deviance explained	22.0%	33.8%	34.9%	52.4%	42.9%	28.0%	31.0%	45.3%	39.1%	27.5%

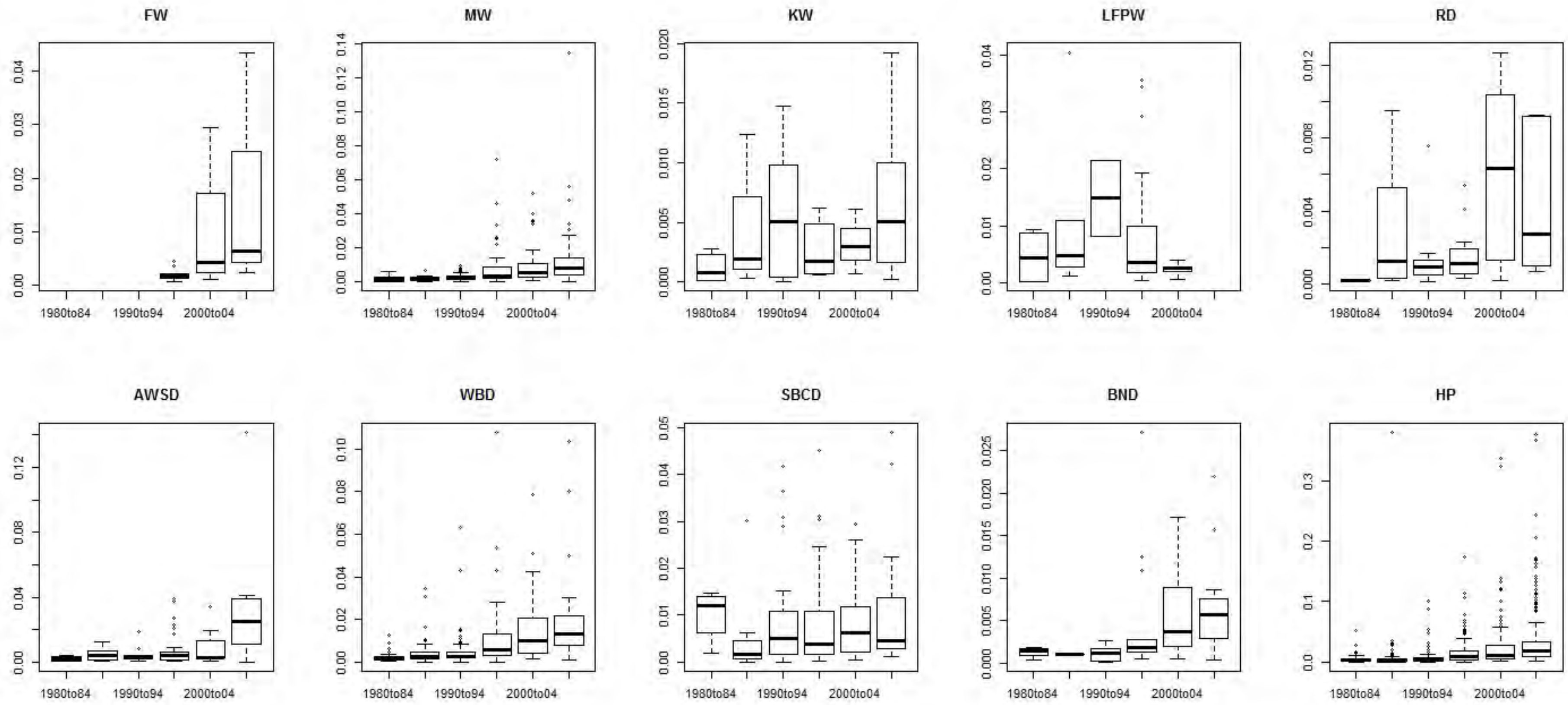


Fig. 18. Box Plots of Sighting Rates with Seismic Shots only, excluding cells with zeros

3.5 Other Human Activities

3.5.1 Hunting

Whales have been hunted in North-west Europe throughout historical times. However, commercial whaling began in the late nineteenth century with the most intense period occurring in the first half of the twentieth century. Eight species have been the target of whale fisheries in the eastern North Atlantic: fin whale, sei whale, blue whale, minke whale, humpback whale, northern right whale, northern bottlenose whale, and long-finned pilot whale. Known catches for each of these species are summarised below. Around the British Isles, whale fisheries operated in the early part of the twentieth century from Shetland and the Outer Hebrides (1903-14, 1918-27, 1950-51) and western Ireland (1908-14, 1920, 1922). Catches given below for the Scottish whale fisheries derive from Thompson (1928) and Brown (1976), and for the Irish whale fishery from Fairley (1981).

Between 1903 and 1928, Scottish catches of fin whales amounted to 4,536 (Shetland) and 1,492 (Outer Hebrides) with a further 46 caught in the Outer Hebrides in 1950-51. Irish catches totalled 435 fin whales between 1908-14 and 157 in 1920 and 1922. Those catches almost certainly depleted the local stocks, the species becoming scarce in the region from then onwards. Most catches occurred off the edge of the continental shelf, particularly north and west of the Shetland Isles, although until the mid-1980's, whaling in Spanish waters may have affected animals occurring off Southwest Britain and Ireland.

Scottish catches of sei whales totalled 1,839 (Shetland) and 375 (Outer Hebrides) between 1903-28, and three in 1950-51 (Outer Hebrides). In W Ireland, 88 were caught between 1908-14, and a further three in the years 1920 and 1922.

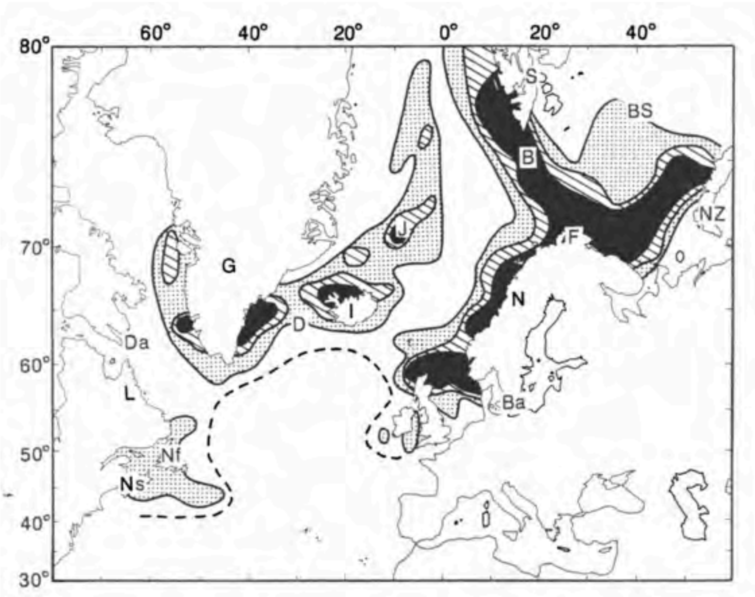


Fig. 19. Distribution of Norwegian catches of minke whales over about 6.1 m, accumulated from 1960 to 1972 in rectangles of 1.5^o longitude and 2.5^o latitude. Dashed line shows southern extent of the whaling contours at 10, 50, and 100: Ns, Nova Scotia; Nf, Newfoundland; L, Labrador; Da, Davis Strait; G, Greenland; D, Denmark Strait; I, Iceland; J, Jan Mayen; S, Spitzbergen; B, Bear sland; BS, Barents Sea; NZ, Novaya Zemlya; F, Finnmark; N, Nordland; Ba, Baltic Sea (Source: Horwood, 1987)

The Scottish whale fishery took 85 blue whales between 1903-28 from Shetland and 310 from the Outer Hebrides. In 1950-51, a further six were captured in Outer Hebridean waters. The Irish whale fishery captured 98 between 1908-14 and 27 in 1920 and 1922. Most captures were made in deep waters off the edge of the continental shelf.

Because of its relatively small size, the minke whale was not a target of the Scottish and Irish whale fisheries in the early years of the twentieth century. In the 1940s, however, minke whales started to be taken along the Scottish and English east coasts, the Scottish whaling taking place in July and August to the east of the Shetland Islands, and then offshore of the English coast in September and October (Horwood, 1990). The main areas where animals were taken is shown in Fig. 19.

Catches of humpback whales in the Scottish whale fishery amounted to 51 (Shetland) and 19 (Outer Hebrides) between 1903-28 and none in the Outer Hebrides between 1950-51. In NW Ireland, six were taken between 1908 and 1914, but with none in 1920 and 1922.

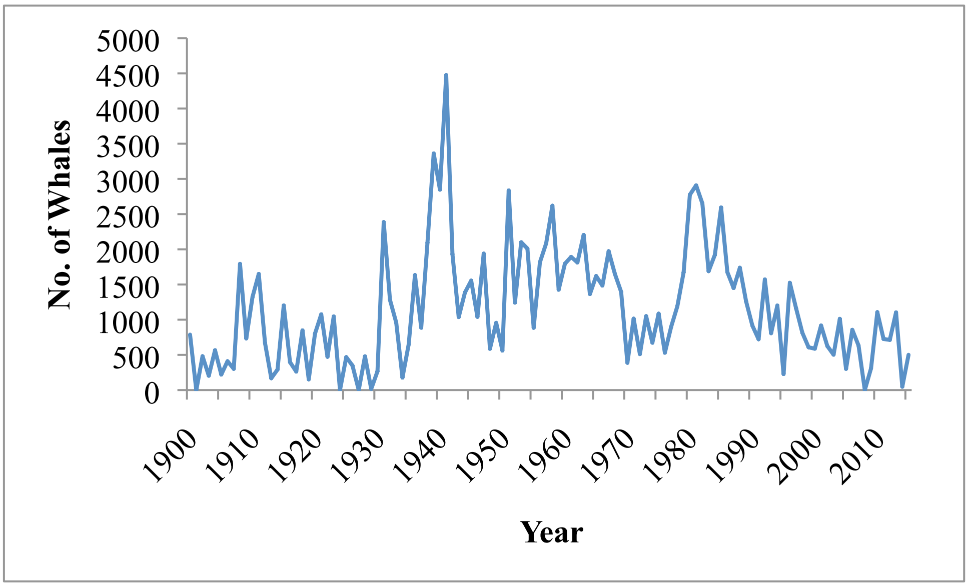


Fig. 20. Catches of Long-finned Pilot Whales in the Faroe Islands, 1900-2015 (Source: Faroese government data)

Between 1903-28, the Scottish whale fishery took 94 northern right whales in the Outer Hebrides, and six in Shetland. Only three were taken between 1918-27 and none when whaling resumed between 1950-51. In W Ireland, 18 were caught between 1908-14 but none in 1920 and 1922.

The Scottish whale fishery took 76 sperm whales in the Outer Hebrides and 19 in Shetland between 1903-29. One individual was taken in the former region in 1950-51. In Western Ireland, 48 sperm whales were taken between 1908-14 and a further 15 in 1920 and 1922. Most catches occurred in deep waters just off the edge of the continental shelf. More recent whaling for this species in the North Atlantic occurred around Iceland, Spain, Madeira and Azores. The last catches were made in 1987.

There have been two main periods of exploitation of northern bottlenose whales in the North Atlantic: about 50,000 were taken in the period 1882-1914 (Holt, 1977), and 5,000 between 1955-72 (Jonsgård, 1977). Small numbers were taken in the Scottish whaling industry early this century, although preference was given to the larger rorquals. Between 1903 and 1928, a total of 25 were captured around

Shetland and one in the Outer Hebrides (none in 1950-51). Most captures occurred in deep waters off the edge of the continental shelf. None was taken off Western Ireland. Between 1938-72, 5,800 animals were taken by Norwegian whalers with the great majority from 1955-72 (Holt, 1977; Jonsgård, 1977).

Organised drives of long-finned pilot whales have taken place for at least eleven centuries in the Faroe Islands, where they continue to the present day with an average annual catch of 850 from 1709 to 1992 (Zachariassen, 1993), and 708 from 1993 to 2015 (Faroese government data; see Fig. 20). Other drive fisheries operated in an opportunistic manner in Britain and Ireland until the early part of the twentieth century, mainly in Shetland and Orkney, but also in the Outer Hebrides and Western Ireland.

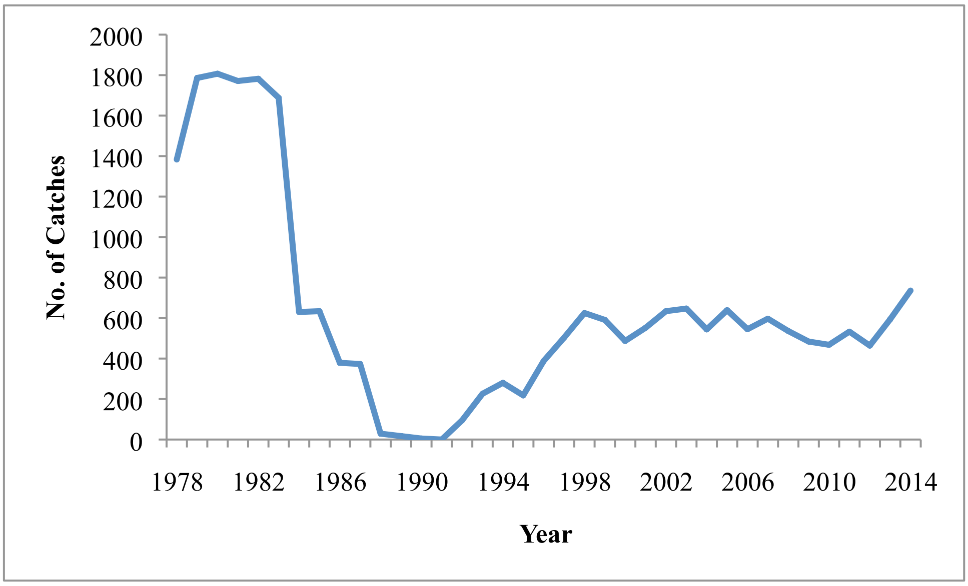


Fig. 21. Minke Whale Catches by Norway in the NE Atlantic, 1978-2014 (source: IWC)

Calculations of pre-exploitation population sizes have been fraught with difficulties. The often wildly different estimates obtained may be due to: inaccuracies in the catch record, uncertainties surrounding genetic estimates, and/or differences in time scales applied to the estimates (Roman and Palumbi, 2003; Holt and Mitchell, 2004; Punt *et al*., 2006; Alter and Palumbi, 2009; Smith and Reeves, 2010; Ruegg *et al*., 2013). However, they generally indicate marked reductions in population sizes for most of the baleen whale species since commercial whaling started. Since the moratorium on commercial whaling was imposed by the IWC in 1986, whaling has largely ceased and populations of some species appear to be recovering, a notable example being the humpback whale (Zerbini *et al*., 2010).

The minke whale is still exploited in the region by Norway, under objection of the IWC, resulting in c. 24,300 animals taken since 1978 (see Fig. 21). However, there is no indication of a decline in numbers of minke whales in NW European seas since the mid 1980s, and indeed there may have been an increase (Schweder *et al*., 1997; Evans *et al*., 2003; Skaug *et al*., 2004; Bøthun *et al*., 2009; Hammond *et al*., 2013; Paxton *et al*., 2016).

3.5.2 Prey depletion from Fishing

Over the last fifty years there have been major changes to the stock sizes of a number of commercial species, due to a combination of fishing pressure and environmental factors. Many stocks are being exploited at levels that are unsustainable (Fig. 22), while the status of a large number of stocks cannot be fully assessed because of inadequate data (Fig. 23) (OSPAR, 2010).

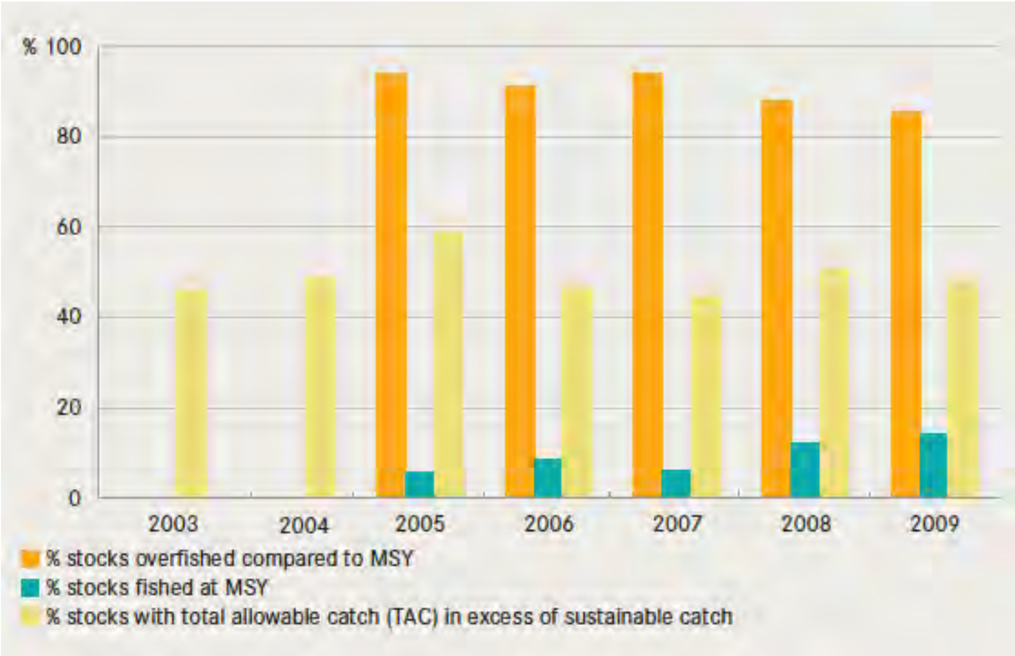


Fig. 22. Status of NW European fish stocks assessed by ICES for which maximum sustainable yield (MSY) is defined. This equates to 32-35 stocks over the period 2005 to 2009, except for 2006 when 23 stocks were assessed on this basis. MSY was not used in fisheries advice before 2005. ICES advice covers over 135 separate fish and shellfish stocks. Source: ICES data. (taken from OSPAR, 2010)

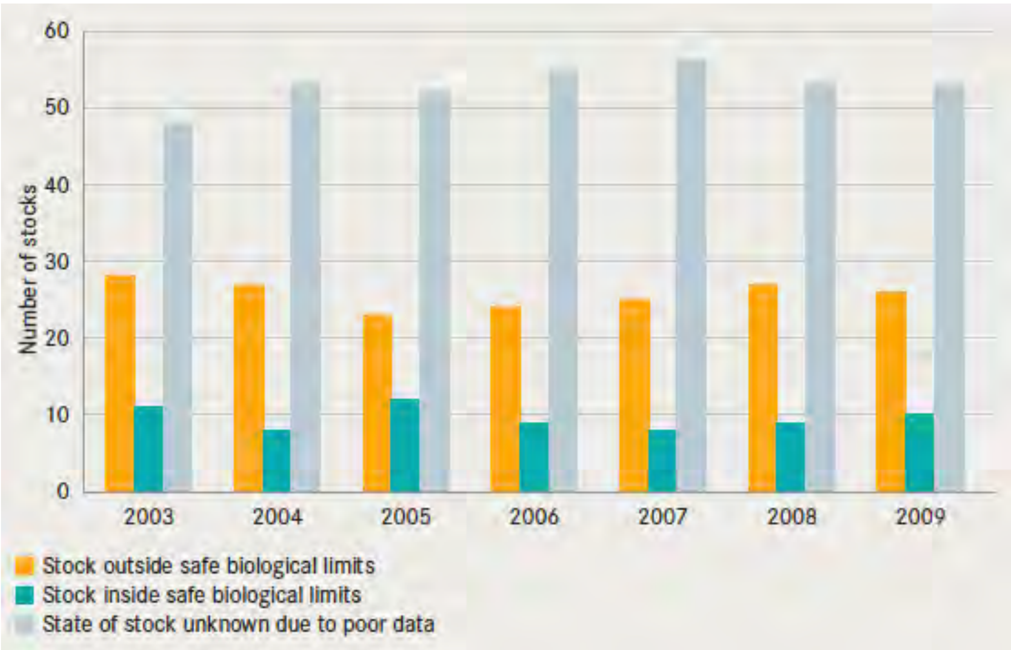


Fig. 23. Status of ICES assessed NW European fish stocks (excluding those in the Baltic Sea) for the period 2003 to 2009. Source: ICES data. (taken from OSPAR, 2010)

The status of around 130 commercial fish stocks in North-west Europe are assessed annually by ICES, as a basis for advice to fisheries authorities on the management of fishing. Individual fish stocks are assessed in terms of spawning stock biomass (SSB), representing the total weight of fish in the stock able to spawn. However, some 48-56 stocks could not be assessed over the period 2003-09 due to poor data (OSPAR, 2010).

Table 6. Feeding ecology and main diet of 18 cetacean species in NW Europe

Species	Foraging Method	Prey species commonly taken
Harbour porpoise	Mainly benthic	Whiting, sandeel, sprat, herring, cod, gobies, pouts
Bottlenose dolphin	Meso- and benthopelagic	Sea bass, salmon, whiting, cod, herring, sandeel, sprat, saithe, haddock, pouts, hake, scad, mullets
Common dolphin	Pelagic	Mackerel, pouts, sardine, anchovy, whiting, scad, sprat, sandeel, blue whiting
Risso's dolphin	Mainly benthic	Octopus, cuttlefish, various small squids
Striped dolphin	Meso- and benthopelagic	Sprat, blue whiting, whiting, silvery pout, pouts, hake, scad, anchovy, bogue, garfish, haddock, saithe, myctophids, gobies, squids
Atlantic white-sided dolphin	Pelagic	Herring, mackerel, silvery pout, blue whiting, scad, argentine, myctophids, squids
White-beaked dolphin	Pelagic	Cod, whiting, herring, mackerel, hake, scad, sprat, pouts, sandeel, haddock, sole, gobies, octopus
Killer whale	Pelagic	Mackerel, herring, salmon, cod, halibut, other marine mammals
Long-finned pilot whale	Benthic and pelagic	Mainly squids; also mackerel, cod, whiting, pollack, scad, sea bass, hake, sole, pouts, eels
Northern bottlenose whale	Benthic and pelagic	Mainly squids (particularly *Gonatus*); also herring, redfish
Sowerby's beaked whale	Mesopelagic	Squids, cod, hake, sandeeel
Blainville's beaked whale	Meso- and benthopelagic	Mainly squids; also gadoids and myctophids
Cuvier's beaked whale	Mainly benthic	Mainly squids; also blue whiting and gadoids
Pygmy sperm whale	Mesopelagic	Mainly squids; some fish and crustaceans
Minke whale	Meso- and benthopelagic	Sandeel, sprat, herring, cod, haddock, saithe, whiting, mackerel, pouts, gobies
Fin whale	Pelagic	Mainly euphausiids, also copepods; herring, mackerel, sandeel, blue whiting, squids
Sei whale	Pelagic	Mainly copepods; also euphausiids, small schooling fishes and squids
Humpback whale	Pelagic	Mainly euphausiids; also herring, sprat, sandeel

A summary of the main prey species taken by cetaceans in NW Europe is given in Table 6. This is based largely upon stomach contents analysis of stranded and by-caught specimens from various localities within the region. No attempt is made here to split by area since, for most cetacean species, prey analyses have been limited to only a few localities. Nevertheless, this does indicate the range of prey species commonly taken, and highlights the overlap with several species targeted by commercial fisheries.

Six commercial fish species occur regularly in the diets of several cetaceans: herring is an important dietary component for nine cetacean species, sandeel for eight species, sprat, mackerel, cod, and whiting for seven species. Trends in spawning stock biomasses for those six fish species are presented in Figures 25 – 30. ICES recently have revised the stock size assessments for herring and whiting, suggesting higher abundance in the North Sea than thought previously. Nevertheless, the overall trends remain the same. Herring stocks collapsed in the mid-1960s and following protection, increased again from the mid-1980s, particularly from around 2000 onwards (Fig. 25). Whiting abundance fell since the 1980s (Fig. 26) and sandeel stocks collapsed throughout much of the North Sea from 2000 (Fig. 27). Sprat (Fig. 28), cod (Fig, 29) and mackerel (Fig. 30) stocks are also much reduced since around 1970, although there has been some recovery since c. 2010, particularly in sprat and mackerel stocks.

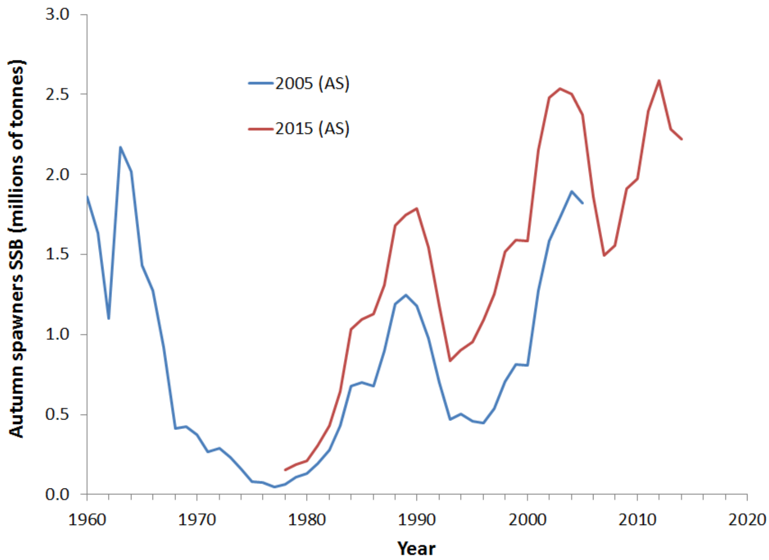


Fig. 24. Estimated spawning stock biomass of (autumn spawning) herring in the North Sea based on assessments made in 2005 and 2015 (Source: ICES)

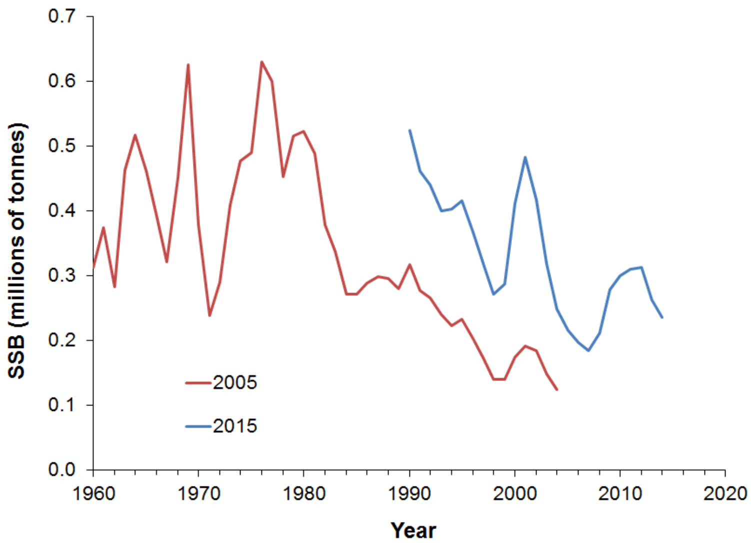


Fig. 25. Estimated spawning stock biomass of whiting in the North Sea based on assessments made in 2005 and 2015 (Source: ICES data)

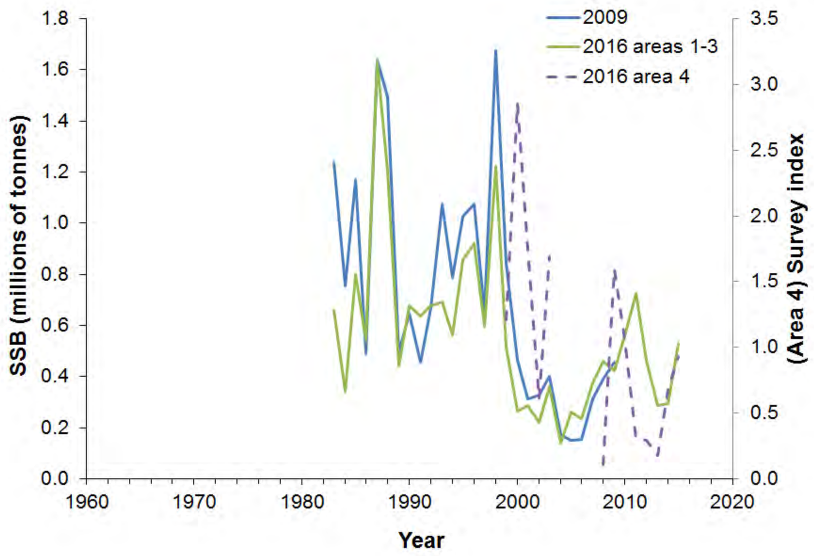


Fig. 26. Estimated spawning stock biomass of sandeel in the North Sea.
Area 1 = W Central N Sea; Area 2 = Southernmost N Sea; Area 3 = E Central N Sea;
Area 4 = NW N Sea (Source: ICES data)

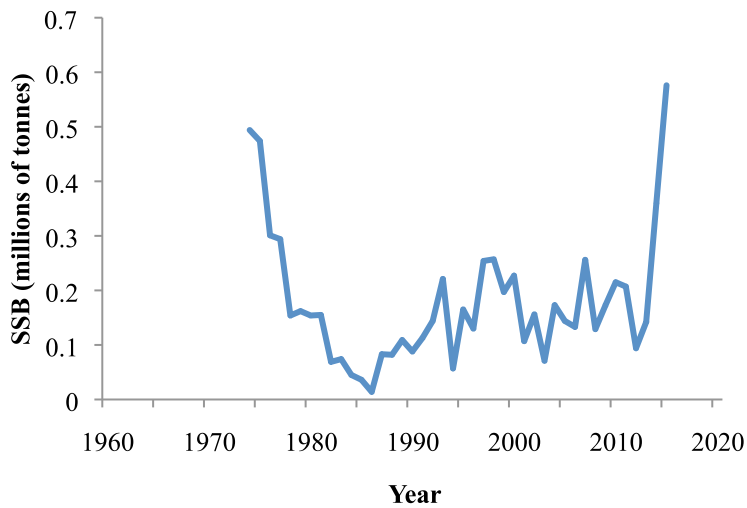


Fig. 27. Estimated spawning stock biomass of sprat in the North Sea.
(Source: ICES data)

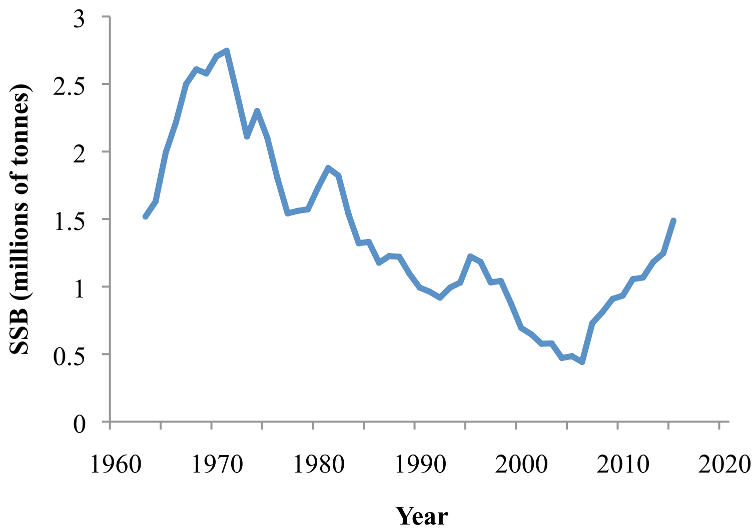


Fig. 28. Estimated spawning stock biomass of cod in the North Sea (Source: ICES data)

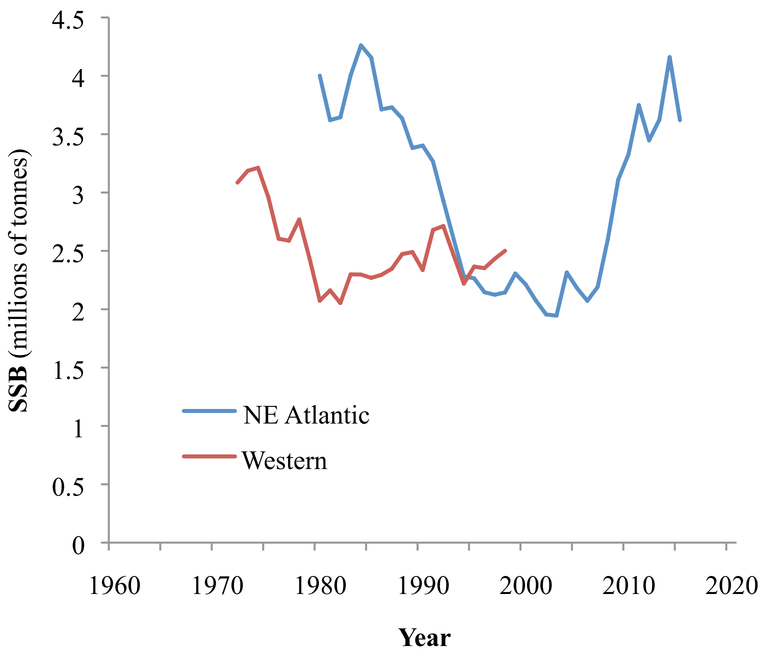


Fig. 29. Estimated spawning stock biomass of mackerel in the NE Atlantic (Source: ICES data)

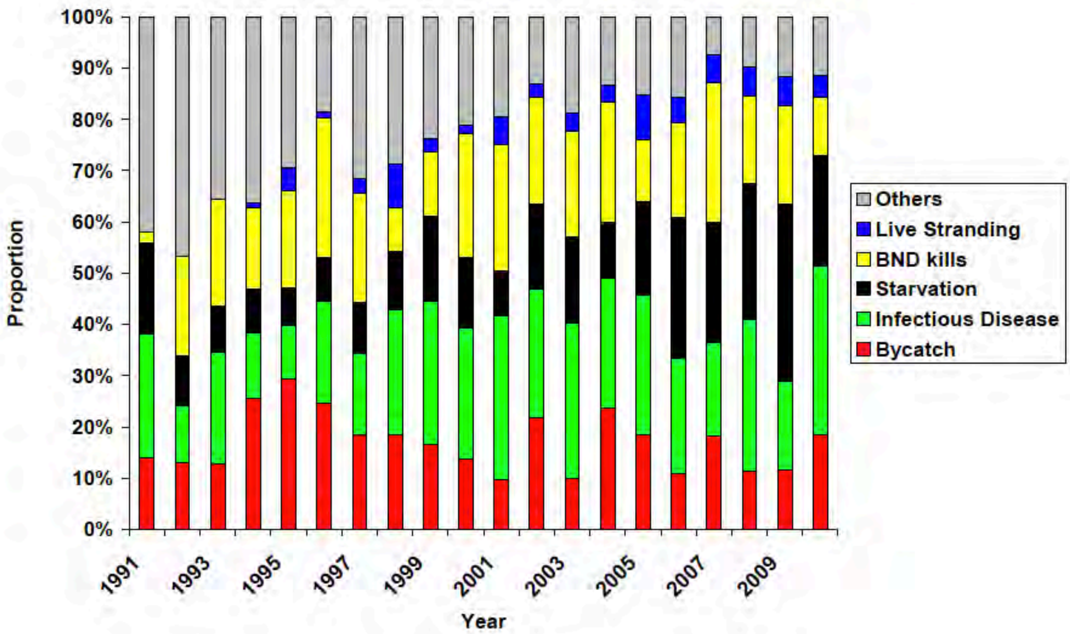


Fig. 30. Annual proportions of main cause of death categories in UK stranded harbour porpoises examined at post-mortem (1991-2010) (from Deaville and Jepson, 2011)

Whereas a cetacean species may respond to reduced availability of a particular fish species by switching prey to another species, Figure 29 shows that >80% of stocks of all commercial fish species assessed by ICES are considered to be overfished in relation to their maximum sustainable yield. This suggests that fishing pressure may therefore have a negative impact on cetaceans affecting their status as well as distribution, as implicated earlier for the harbour porpoise by Evans (1990) and Reijnders (1992). As yet, we cannot establish whether there is a population level effect through lower energy intake resulting in reduced fecundity or survival. However, of some concern is the general rise in incidence of starvation amongst post-mortem examinations of harbour porpoise from the UK (Deaville and Jepson, 2013; see Fig. 30).

3.5.3 Incidental Mortality in Fishing Gear

The most obvious direct effect of human activities upon cetaceans in North-west Europe is mortality from entanglement in fishing gear. Virtually every cetacean species in the world has been known to die from accidental capture in fishing gear (Northridge and Hofman, 1999). Likewise, almost all kinds of fishing operations have at least some impact on cetaceans, and some interactions represent a significant threat to them globally (Reeves *et al*., 2003; Read *et al*., 2006; Reeves *et al*., 2013). There is almost no information on the history of such interactions before the early 1970s (Reeves *et al*., 2003). One may assume that by-catch has existed long before and indeed there are reports of porpoise by-catch in the early part of the 20th century in the southern North Sea herring fishery (Evans and Scanlan, 1989). Since the middle of the last century, modern technology such as fish-finders and sonar, have made detection of shoals relatively easy. Synthetic materials, including monofilament fibres for netting, have lessened the chances of breakage or escape once the fish are caught. These improved techniques are thought to have led to a resultant increase in by-catch (Northridge, 2009).

Although almost any gear can cause entanglement, certain types are known to be more problematic, and may affect particular species more than others (see Table. 7).

Table 7. Species / Gear Interactions - fishing gear known to cause accidental entanglement for major European cetacean species (adapted from Northridge, 2009)

Species/Gear category	Gill nets	Pelagic trawls	Demersal trawls	Long lines	Drift nets	Seine nets	Pot lines
Harbour porpoise	√		√		√		
Bottlenose dolphin	√	√	√				√
Atlantic white-sided dolphin	√	√			√		
White-beaked dolphin	√	√					
Short-beaked common dolphin	√	√	√		√	√	
Striped dolphin	√	√	√		√	√	
Risso's dolphin				√			
Killer whale				√			
Long-finned pilot whale	√	√	√	√			
Minke whale	√	√					√
Fin whale							√
Humpback whale							√

NOTE: Current sampling based on frequency of records, not necessarily the significance of possible impact

In North-west Europe, five types of fishing gear, which have been operated over the last two decades, are particularly identified as having a cetacean by-catch associated with them. These are: gill nets, pelagic trawls, driftnets, seine nets, and pot lines (Northridge and Hofman, 1999; Kaschner, 2003; Northridge, 2009; Northridge *et al*, 2010; Evans and Hintner, 2012; Brown *et al*., 2013).

Two species above all other cetacean species in North-west Europe, appear to be major victims of by-catch from **gillnet fisheries**. These are the harbour porpoise and short-beaked common dolphin. Gillnets, as well as tangle nets and trammel nets (other forms of gillnet) are all deployed on or near the seabed, targeting demersal fish species. Probably due to harbour porpoise feeding behaviour on or near the seabed, those gear types are associated with having the highest harbour porpoise mortalities (Northridge, 1988, 1991; Northridge and Hammond, 1999; Northridge and Hofman, 1999). During the 1990s, annual catches in bottom set gillnets in the North Sea were estimated at c. 8,000 porpoises (Vinther, 1995, 1999; Northridge and Hammond, 1999), i.e. 3% of the estimated abundance of c. 250,000 (Hammond *et al*., 2002), and almost twice the threshold limit set by ASCOBANS as unsustainable.

Although all countries bordering the North Sea and adjacent waters have reported by-catch in their fisheries, the largest fishery in the region is the Danish bottom-set gillnet and wreck net fisheries. For the years up to 2001-02, Vinther and Larsen (2004) estimated an annual by-catch of 5,591-5,817 porpoises from the central and southern North Sea. The former figure is based on landings as used by Vinther (1999), and the latter is extrapolated from by-catch rates determined from observers between 1987 and 2001, accounting for fleet effort. By-catch may have been overestimated due to use of pingers (acoustic deterrent devices) in the cod wreck net fishery not being accounted for (Vinther and Larsen, 2004).

The UK and Irish hake/pollack gillnet fishery in the western English Channel and Approaches (referred to as the Celtic Sea) also had a significant annual by-catch of harbour porpoises, estimated for 1992-94 at 2,200 (with c. 700 in the UK and 1,500 in the Irish fisheries) (Tregenza and Hammond, 1994; Tregenza *et al*., 1997a). This represented 6.2% of the estimated number of porpoises in the region, a level more than three times the amount that was considered sustainable (Tregenza *et al*., 1997a; Hammond *et al*., 2002). In this same fishery, common dolphin by-catch rates were estimated in this fishery over the years 1992-94 and found to be around 200 animals per year (Tregenza *et al*., 1997b; Tregenza and Collet, 1998). The annual by-catch of common dolphins in Irish gill net fisheries for hake and cod in the Celtic Sea between 2006 and 2007 was approximately double what it had been in 1992–1994, and may have underestimated overall by-catch since all common dolphins recorded in the earlier period were caught in late autumn and winter, a period that was not sampled in the later study (Tregenza *et al*., 1997b, Cosgrove and Browne 2007).

Since around 2000, estimated by-catch levels of both harbour porpoise and common dolphin have appeared to decline (see Table 8), due probably to subsequent management action and a decline in overall fishing effort. However, for most years, by-catch still figures highly as a cause of death amongst common dolphin post-mortem examinations in the UK (Deaville and Jepson, 2011; Deaville, 2015; Figs. 32 and 33), and bycatch levels in the French pelagic trawl fishery in the Celtic Sea may be unsustainable (ICES, 2016). There is also indication that harbour porpoise bycatch in small vessel fisheries in Norwegian coastal waters is significant (Bjørge *et al*., 2013). Using data collected during 2006–2008 from a monitored segment (18 vessels) of the Norwegian coastal fleet (vessels <15 m) of gillnetters targeting monkfish and cod, Bjørge *et al*. (2013) estimated that c. 6,900 harbour porpoises are taken annually by the coastal gillnet fisheries whilst annual bycatch in the UK gillnet fleet currently numbers around 1,400-1,700 (ICES WGBYC, 2016).

During recent decades, multi-national **pelagic pair trawl** fisheries for bass have operated each winter in the Celtic Sea and western English Channel. The offshore pelagic trawl fishery has been predominantly a French fishery, with about three quarters of annual fishing effort in the western Channel due to French vessels, whilst about a quarter were UK vessels, mainly from Scotland. It was estimated that between 2000 and 2003, the UK fishery in the Channel took around 90 common dolphins annually (but no porpoises) (Northridge *et al*., 2003). However, this likely underestimated the total by-catch since annual strandings of common dolphins alone over that period exceeded 90 every year (Jepson, 2005). More recently, common dolphin by-catch estimates in the UK bass pelagic pair trawl fishery were 84 (2005-06) in ICES Area VIIe, and 114 (2006-07) in ICES Area VIIadefghi (UK National Report to ASCOBANS, 2009). Pooling observation data from 2005 to 2014, common dolphin bycatch in UK set net fisheries was estimated at 276 for the year 2014 (ICES WGBYC, 2016). This heavy bycatch was reflected in the proportion of stranded common dolphins post-mortemed where cause of death was attributabl to bycatch, although there are large fluctuations between years (Figs. 31 and 32)

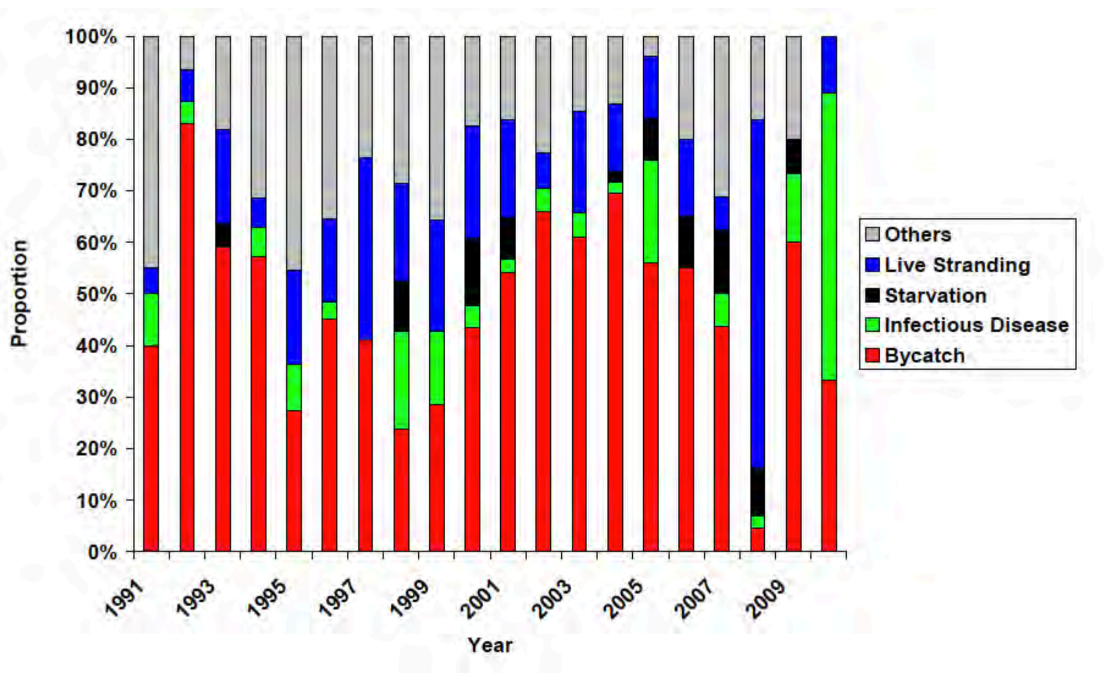


Fig. 31. Annual proportions of main cause of death categories in UK stranded short-beaked common dolphins examined at post-mortem (1991-2010)
(Source: Deaville and Jepson, 2011)

Independent observer schemes targeting the French pelagic trawl fishery in the mid-1990s estimated by-catches of common and striped dolphins between the low hundreds and low thousands per year (Morizur *et al*., 1996, 1999; Tregenza and Collet, 1998). Following the introduction of EC Regulation 812/2004 in 2004, a by-catch of 240 common dolphins, 40 striped dolphins, 50 bottlenose dolphins, and 10 long-finned pilot whales was estimated in pelagic trawls for 2007 (French Annual Report for 2007 to ASCOBANS, 2009), and of 300 common dolphins and 90 long-finned pilot whales in 2008 (French Annual Report to ASCOBANS, 2010).

During the early to mid-1990s, common dolphins were reported as by-catch in Dutch horse mackerel pelagic trawl nets fishing off the SW coast of Ireland and French hake pelagic trawl nets in the inner Bay of Biscay (Couperus 1997a, b, Tregenza and Collet 1998, Morizur *et al*., 1999). The Dutch pelagic trawl fishery for horse mackerel also caught significant numbers of Atlantic white-sided dolphins (Couperus 1997a, b).

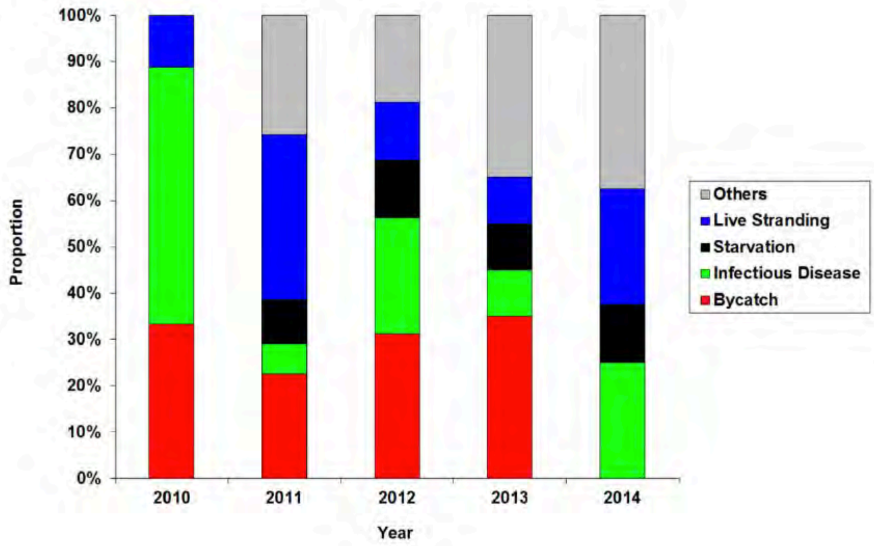


Fig. 32. Annual proportions of main cause of death categories in UK stranded short-beaked common dolphins examined at post-mortem (2010-2014) (Source: Deaville, 2015)

Initial investigations into the Irish pelagic trawl fishery for albacore tuna were carried out in 1996 and 1998, and it was estimated that 345 and 2,552 common dolphins, respectively, were caught incidentally by the whole fishery (Harwood *et al*., 1999). During 1998 and 1999, An Bord Iascaigh Mhara (BIM) and the Marine Institute undertook a major two-year study into developing alternative tuna fishing techniques (BIM, 2004). In 1999, tests on experimental trawls were carried out off Western Ireland and the southern Bay of Biscay, and in 313 hauls over 160 days, a total of 145 animals were by-caught, including 125 common dolphins, 10 striped dolphins, eight long-finned pilot whales and two Atlantic white-sided dolphins, although these were all caught in just four trawls (BIM, 2000). Since 1999, by-catch from this fishery appears to have been much lower, possibly as a result of the management measures introduced (BIM, 2004, 2005; ICES WGBYC, 2012).

Very high vertical opening (VHVO) bottom pair trawl fisheries can also result in cetacean by-catch, with common dolphins in particular recorded in the French and Spanish fisheries operating in the Bay of Biscay and Celtic Sea (López *et al*., 2003; ICES WGMME, 2005; Northridge *et al*., 2006).

There is relatively little **long lining** practised around the British Isles (mainly small and inshore), with the Norwegian, Icelandic and Faroese fisheries dominating the industry, operating on the shelf and shelf edge north of the British Isles (Brothers *et al*., 1999). As a consequence, although cetacean by-catch from long lining can be high in various parts of the world (e.g. Gulf of Mexico), it does not seem to be an issue in the region under consideration here.

During the 1980s, the use of large-scale **driftnets** was established in most oceans of the world, with nets of up to 50km in length regularly deployed in the Pacific. They resulted in very sizeable by-catches in many regions (IWC, 1994), including not only cetaceans but also seabirds, turtles, sharks and other non-target fish species.

In Europe, there were major driftnet fisheries in the eastern North Atlantic for tuna (French and Spanish fisheries), for small pelagic fish in the Mediterranean and central Baltic, and along the Atlantic coasts of Norway and Ireland (as well as off West Greenland) for salmon (IWC, 1994). The principal species by-caught were harbour porpoises near-shore, and common, striped and Atlantic white-sided dolphins offshore. An independent observer scheme targeting French tuna driftnet fisheries in the Celtic Shelf and Bay of Biscay during 1992-93 estimated by-catch of mainly striped dolphins to be between one and two thousand per year (Goujon *et al*., 1993; Goujon, 1996). Using landings of albacore tuna as an indicator of effort, an overall by-catch of 11,723 (CI = 7,670–15,776) common dolphins was estimated for the period 1990 to 2000 (Rogan and Mackey, 2007).

Following widespread concerns over high by-catches reported globally, the United Nations imposed a moratorium on the use of all large-scale pelagic driftnets by 30 June 1992, and the European Community (EC) responded with a series of resolutions leading to a total ban on the use of driftnets in Atlantic waters, which came into force at the start of 2002.

Incidental capture from purse **seine netting** for tuna was a major conservation problem for a number of dolphin species in the eastern tropical Pacific during the 1980s and 1990s (Hall, 1998; Northridge and Hofman, 1999; Hall and Donovan, 2001). Although clearly having the potential to be damaging to dolphin populations, this particular mode of fishing (with seine nets set around pods of dolphins) is no longer used in the North Atlantic. Most other seine netting in North-west Europe occurs in the northern North Sea east of Scotland, but there is a small amount in the eastern Channel and the northern Irish Sea between the Isle of Man and coast of Co. Dublin (Evans and Hintner, 2010). These do not appear to have a by-catch associated, although killer whales are well known to depredate herring and mackerel when the seine nets are being hauled (Couperus, 1993, 1994; Luque *et al*., 2006).

The setting of **pots** or **traps** for fish or crustaceans can also entangle cetaceans. They often become caught in the leader ropes rather than the traps themselves, and amongst the more commonly caught species are baleen whales, such as the humpback and minke whale (Lien, 1994; Lien *et al.,* 1995; Northridge *et al*., 2010; Ryan *et al*., 2016). Most entanglements in North-west Europe have been reported from north and west Scotland, involving minke whales and humpbacks (Northridge *et al*., 2010; Ryan *et al*., 2016).

Finally, one should mention ghost netting. Fishing nets and lines that are cut loose and discarded may entangle cetaceans. Around the British Isles, there have been several cases involving minke whales, harbour porpoises, and grey seals amongst other species being found entangled in lost/discarded gear (Northridge, 1988; Evans, 1993; Evans and Hintner, 2010).

Table 8. Summary of Fisheries and By-catch Information for North West Europe

Area (and ICES area if known)	Gear type	Target species	Year	Species	By-catch levels	Estimated Mean Annual By-catch	Source	By-catch Investigation approach and Comments
Irish Sea VIIIa-e, VIIh,j,k	Driftnet	Albacore Tuna	1995	CD, SD	Medium	Low 100s	CEC, 2002b	Monitoring scheme By-catch decline with low effort, fishery terminated by EC regs. in 2002
North Sea (offshore) IIa,Iva,Ivb,IVc	Static	Cod, skate, turbot, sole, monkfish, dogfish	1995-1999	HP	High	100s	CEC 2002a,b: Defra, 2001; Northridge & Hammond, 1999; SFPA / SFI, 2001	Monitoring scheme By catch estimate without freezer-netter fleet
North Sea (inshore) Iia,Iva,Ivb,IVc	Static	Cod	1995-1999	HP	Medium	100s	CEC, 2002a, b; Defra, 2001; Northridge & Hammond, 1999; SFPA/SFI, 2001	Monitoring scheme Bycatch estimate without freezer-netter fleet
West of Scotland Via	Static	Dogfish, crayfish, skate	1995-1999	HP, CD	Medium	Low 100s	Northridge, in CEC, 2002a	Monitoring scheme Drastic decline due to collapse of crayfish fishery

Channel VIId,e	Static	Cod, monkfish, flatfish	-	HP	Low?	-	ASCOBANS, 2003a; CEC, 2002a,b	Opportunistic records
Celtic Sea VIIf-j	Static	Hake, cod, pollack, saithe, ling	1992-1994	HP, CD	Medium-high	100s	CEC 2002a,b: Tregenza *et al.*, 1997; Tregenza & Collet, 1998	Monitoring scheme
Bay of Biscay, Celtic Shelf VIIg-k	Pelagic pair trawl	Albacore tuna	2000-2010	Mainly CD, also SD, AWSD, WBD, LFPW	High?	10s to 100s	CEC, 2002b; ICES, 2008; Y. Morizur *pers. comm.*	Monitoring scheme
North Sea and West of Ireland IVa-c, Via,b	Pelagic trawl	Herring, mackerel	1995-1996 and 2000-2001	LFPW, potentially other species	Low?	-	ASCOBANS, 2003a;CEC, 2002a,b; Morizur *et al.*, 1999	Monitoring scheme
Western Channel VIId,e	Pelagic pair trawl	Mackerel, bass, pilchard, blue whiting, and anchovy	1995-1996 and 2000-2001	CD, SD, AWSD, WBD, LFPW	High, mainly CD	-	CEC, 2002b; Morizur *et al.*, 1999	Monitoring scheme
North Sea and ? IVb,c and others?	Demersal trawl	Cod and others?	-	HP	Very low?	-	CEC, 2002b	NONE
Northern North Sea IIa, Iva (parts)	Purse seine	Herring, mackerel	-	Small cetaceans	Low?	-	CEC, 2002b	Opportunistic records
North Sea IVa, IVb, IVc	Fish trap	Salmonids	-	HP	Low?	-	CEC, 2002b	NONE

North Sea IV	Set nets	Cod, skate, turbot, sole, monkfish	1995-2002	HP	Medium	439 [371-640]	ASCOBANS, 2004	NONE
North Sea IV	Set nets	Cod, turbot, sole, other demersal fish	2002-2003	HP		25-30	Flores & Kock, 2003	Independent observer scheme
North Sea IV, VIID, IIIA	Set nets		2012-2014	HP		27-29/1000 days at sea	ICES WGBYC, 2015	Remote Electronic Monitoring
North Sea including VIId and IIIa	Set nets		2013-2014	HP	High	1235-1990	ICES WGBYC, 2015	Independent observer scheme
English Channel, Celtic Sea and North Sea	Gill nets and trammel nets		2013	HP	High	1600-1900	ICES WGBYC, 2015	Independent observer scheme
English Channel, Celtic Sea and North Sea	Gill nets and trammel nets		2014	HP	High	1400-1700	ICES WGBYC, 2016	Independent observer scheme
Channel and Bay of Biscay VIId,e,f, VIIIa,b and some in IVc	Fixed	Sole, anglerfish, cod, hake, turbot	1995-1996	HP	Low?	<1	ASCOBANS, 2003c; Morizur *et al.,* 1996; CEC, 2002b	
Channel VIId,e	Fixed	?	-	HP	Medium?	>10	Morizur *et al.*, 1996; Swarbrick e*t al*., 1994	1 HP per boat per year (potentially up to 30 boats)
Celtic Sea VIIe-j	Fixed	Hake and anglerfish	?	HP and other species	High?	-	Morizur *pers. comm.,* in CEC, 2002b	

North Sea VIa,b	Pelagic single or pair trawl	Herring, mackerel and horse mackerel	-	HP, LFPW and small cetaceans	Very low?	-	ASCOBANS, 2003c; CEC, 2002b	NONE
Celtic and Irish Seas			2012-2014	HP	High	1137-1472	ICES WGBYC, 2015	Independent Observer Scheme
Western Channel (and Celtic Shelf?)	Pelagic single or pair trawl	Blue whiting, mackerel and horse mackerel, herring, sea bass, black sea bream	1994-1995	CD, AWSD, and other species	High for all species but mainly CD	100s	ASCOBANS, 2003c; CEC, 2002a,b; Morizur *et al.*, 1996, 1999	Independent Observer Scheme
Celtic Shelf and Bay of Biscay VIIIa, b, d	Pelagic single or pair trawl	Hake, tuna, sardine, anchovy, horse mackerel, sea bass	1994-1995	CD, BND	High for all species but mainly CD	100s	ASCOBANS, 2003c; CEC, 2002a,b; Morizur *et al.*, 1996, 1999	Independent Observer Scheme
Celtic Shelf and Bay of Biscay VIIIa, b, d	Pelagic single or pair trawl	Manly sea bass	2000-2010	Manly CD	High	Up to 1,000 (2009)	ICES, 2008; Y. Moriizur *pers. comm.*.	Independent Observer Scheme
English Channel and Bay of Biscay	Set nets, mainly trammel nets	Monkfish, turbot and sole	2008-2013	HP	High	600	Morizur *et al.*, 2014; ICES WGBYC, 2015	Independent Observer Scheme
Celtic Shelf and Bay of Biscay VIIIa, b, d	Pelagic single or pair trawl, set net, and purse seine		2008-2013	CD	High	2509	ICES WGBYC, 2015, ICES, 2016	Independent Observer Scheme

Notes:

Key to species

Harbour porpoise	HP
Common dolphin	CD
Bottlenose dolphin	BND
Striped dolphin	SD
Atlantic white-sided dolphin	AWSD
Minke whale	MW
White-beaked dolphin	WBD
Long-finned pilot whale	LFPW

Annual By-catch levels

Rare	Very low
<10/year	Low
10-500 animals/year	Medium
>500 animals/year	High
Several 1000 animals/year	Very high
Potential by-catch levels for fisheries not yet monitored using independent observer programs but alternative sources of information available.	?

3.5.4 Chemical Pollutants and Other Hazardous Substances

Chemicals can be naturally occurring, like metals in the Earth's crust, formed as unintended by-products of natural and human-induced chemical processes, or synthesised specifically for use in industrial processes and consumer products. About 100,000 substances are on the European market and around 30,000 of these have an annual production of more than one tonne per year (OSPAR, 2010). Some of these are hazardous because they are persistent, liable to accumulate in living organisms, and toxic. They can contaminate the marine environment, with harmful effects on marine life. Cetaceans as top predators are particularly vulnerable to high contaminant burdens if those chemicals are persistent (known as POPs – Persistent Organic Pollutants) and can biomagnify levels up the food chain. Examples of these are the chlorinated pesticides like DDT and dieldrin, the polychlorinated biphenyls (PCBs), and various heavy metals such as cadmium and mercury. Other metals can be highly toxic, killing organisms outright. The characteristics and general effects of the main pollutants affecting cetaceans are listed in Table 9.

Table 9. Characteristics of the major pollutants known to negatively affect cetaceans

Pollutant	Characteristics	Effects
Organohalogens - chlorinated pesticides (DDT, dieldrin, endrin, mirex), polychlorinated biphenyls (PCBs), Hexachlorocyclohexane (HCH), polybrominated biphenyls (PBBs), and polybrominated diphenyl ethers (PBDEs), polycyclic aromatic hydrocarbons (PAHs), and phenols	lipid soluble, persistent, and biomagnify up the food chain, and bioaccumulate with age	endocrine impairment, reproductive impairment, and increased susceptibility to disease
Metals - methyl mercury, organotins, lead, cadmium, copper, and zinc	some (e.g. mercury, cadmium) biomagnify up the food chain	can be highly toxic, and cause organ damage
Pathogens - untreated faecal matter from humans, fish farms, etc	can be infectious (bacteria, viruses) or contagious (fungi)	can lead to disease, affecting metabolic systems, altering physiological functions (including reproduction), and causing lesions

Details of the various pollutants and their effects on cetaceans can be found in a number of reviews and volumes of collected papers (see, for example, O'Shea, 1999; Reijnders *et al*., 1999; Hall, 2001; Vos *et al*., 2003; Evans, 2014). In recent years, increasing attention has been paid to brominated flame retardants, and polyaromatic hydrocarbons (Simmonds *et al*., 2002), pefluorinated compounds (Van de Vijver *et al*., 2004) and radionuclides (Watson *et al*., 1999; Nakamura *et al*., 2015).

Pollutants enter the body through the diet, and toxins such as POPs are lipophilic compounds that accumulate in the lipid-rich blubber of cetaceans. During pregnancy in cetaceans, lipid-soluble contaminants, such as organochlorines (OCs), may be transferred from the mother to the foetus. However, the majority (~80% of OCs) of the pollutant burden accumulated by females (primarily prior to sexual maturity), is believed to be transferred to their firstborn calf during the first few weeks of lactation (Cockcroft *et al*., 1989). Thus pollutant levels in females may actually decline with age whereas for males, they progressively increase (O'Shea, 1999; Reijnders *et al*., 1999).

A large number of organochlorine compounds such as DDT and PCBs are hormone- or endocrine-disrupting chemicals. Endocrine functions can be altered by these toxins through interference with the synthesis, secretion, transport, binding, action, or elimination of the endogenous natural hormones responsible for homeostasis, reproduction, development, and behaviour (EPA, 1997).

POPs may also affect the immune system and strong links have been found between elevated blubber PCB levels and mortality from infectious disease (Jepson *et al*., 1999, 2005a; Hall *et al*., 2006) consistent with fatal PCB-induced immunosuppression. In one case-control study of UK–stranded harbour porpoises, the risk of infectious disease mortality increased by 2% for every 1% increase in the summed concentration of 25 CB congeners (Hall *et al*., 2006). A doubling of risk occurred at c. 45 mg/kg (blubber) lipid. In a second case-control study of UK-stranded harbour porpoises, mean summed 25 CB congeners in the 'healthy' control group (death due to physical trauma) was 13.6 mg/kg, compared with 27.6 mg/kg for the animals that died of infectious diseases (Jepson *et al*., 2005a).

Pollutant levels above a threshold of 17 mg/kg PCB lipid weight: mass of PCB per unit mass of lipid are thought to have adverse health effects, based on experimental studies of both immunological and reproductive effects in seals, otters, and mink (Kannan *et al*., 2000).

The first major environmental effects were observed in the 1960s following the widespread use of organochlorine pesticides such as DDT and dieldrin, leading to reproductive impairment in top predators such as raptorial birds. At the same time, PCBs, a by-product of the plastics industry, entered the environment, resulting in a marked increase in levels of this contaminant during the 1960s and 1970s (Fig. 34).

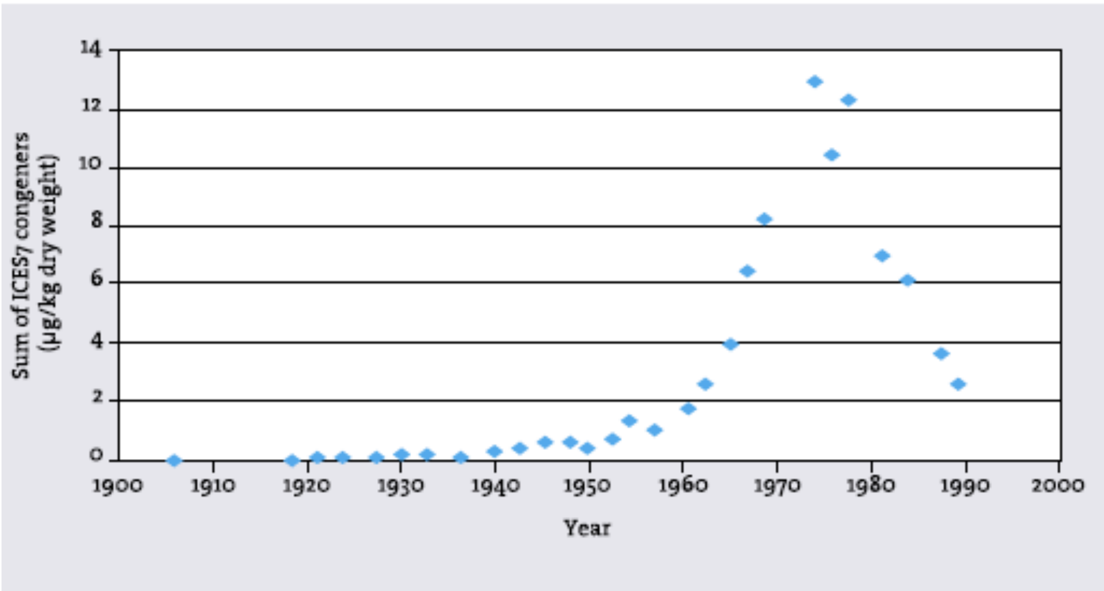


Fig. 33. Trend in PCB concentrations in marine sediments in Eastern England (source: ICES data)

The production of pesticides such as DDT and dieldrin has been completely banned throughout North-west Europe since the 1970s, and for PCBs since the mid-1980s. This has resulted in declines, albeit slow, in levels of those contaminants in both the terrestrial (Loganathan and Kannan, 1994) and marine environment (OSPAR, 2000, 2010; Fig. 33).

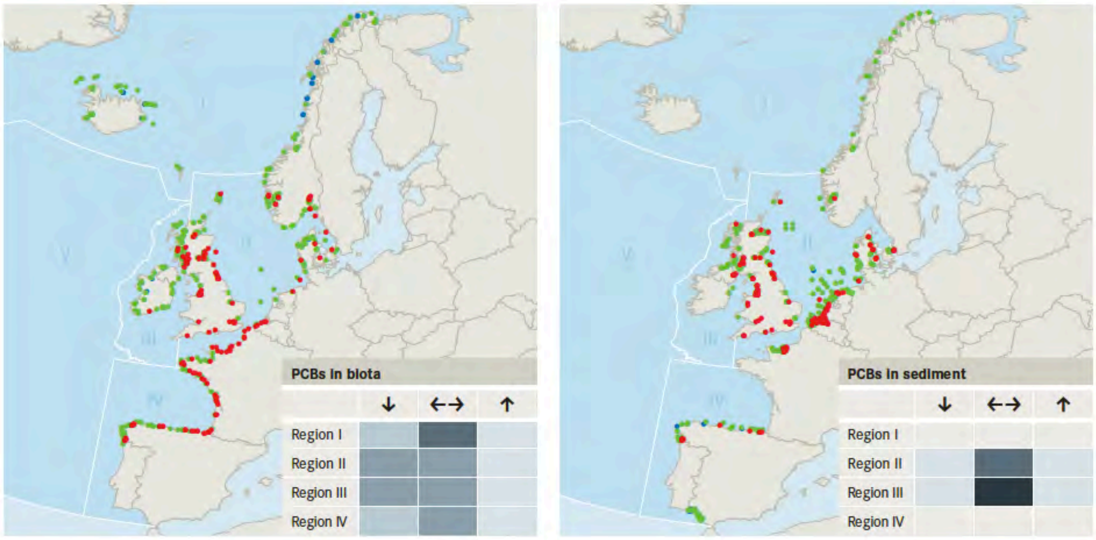


Fig. 34. Geographical distribution of status and temporal trends in contamination from PCBs in biota (fish and shellfish) and sediments based on the OSPAR Coordinated Environmental Monitoring Programme (Source: OSPAR, 2010)

[Status is indicated for the last year of monitoring in the period 2003-2007. Geographic coverage of the assessment is limited, especially for sediments, as a result of lack of data reporting or the design of national monitoring programmes accounting for local conditions. No OSPAR monitoring data have been reported for Region V. Red = unacceptable; Green = acceptable levels]

Total accumulated world production of PCBs has been estimated at 2 million tonnes and much is still contained in sealed systems (OSPAR, 2000). Releases occur for example as leaks from sealed systems (e.g. sealants used in buildings), accidental losses and spills, and emissions from PCB-containing materials and soils. Although OSPAR countries had banned the major PCB uses for some years, and both OSPAR and EU regulations aimed at a complete phase out of PCBs in the period between 1995 and 2010, not all PCBs in smaller applications, in particular in electrical equipment, were likely to be removed within that period (OSPAR, 2000, 2010). Large reductions in the release and phasing-out of remaining stocks were achieved in the period 1998-2005, but releases to air and water are still continuing (OSPAR, 2010). PCBs emitted and deposited during the years of intensive production and use will remain a diffuse source to the global environment. Evaporation of PCBs from polluted soils and waters has been shown to be a significant source to the atmosphere. Once in the atmosphere, PCBs enter the global circulation and can be transported to remote places. The atmospheric input through precipitation in the OSPAR Convention area is estimated to be 3 – 7 t/yr for the period 1992 to 1994 (OSPAR, 2000). Riverine and direct inputs of PCBs are low in absolute terms. Although it is not possible to derive reliable estimates of inputs because most concentrations are below the limit of detection, estimates derived for the Greater North Sea were in the range 0.13 – 2.4 t/yr for the period 1990 to 1995 (OSPAR, 2000).

The results of the OSPAR Coordinated Environmental Monitoring Programme show that levels of PCBs both in sediments and biota remain unacceptably high for many parts of the British Isles and countries bordering the southern North Sea (OSPAR, 2010; Fig. 34).

The monitoring of levels of PCBs in European cetaceans largely started in the 1990s, and very few species have provided sufficient sample sizes to determine trends (Kleivane *et al*., 1995; Berggren *et al*., 1995, 1999; Borrell and Reijnders, 1999; Bruhn *et al*., 1999). The best example is the harbour porpoise where animals stranded in the UK have been analysed for contaminants since 1990 (Kuiken *et al*., 1994; Jepson 2005; Law *et al*., 2006, 2010a, b; Deaville and Jepson, 2011; Law *et al*., 2012a, b, 2013; Jepson *et al*., 2016). These show a significant trend in PCB levels, declining slowly from 1990 and 1998 and then remaining relatively stable from 1998 to 2012 (Jepson *et al*., 2016; Fig. 34). Organochlorine pesticides (such as DDT and dieldrin) showed clearer declines (Law *et al*., 2012a), whilst PBDEs (penta-mix brominated diphenyl ether congeners), after an initial increase in the late 1990s, have also declined (Law *et al*., 2010). Another POP, the butyltins (including TBT), showed only trace levels (Law *et al*., 2012b).

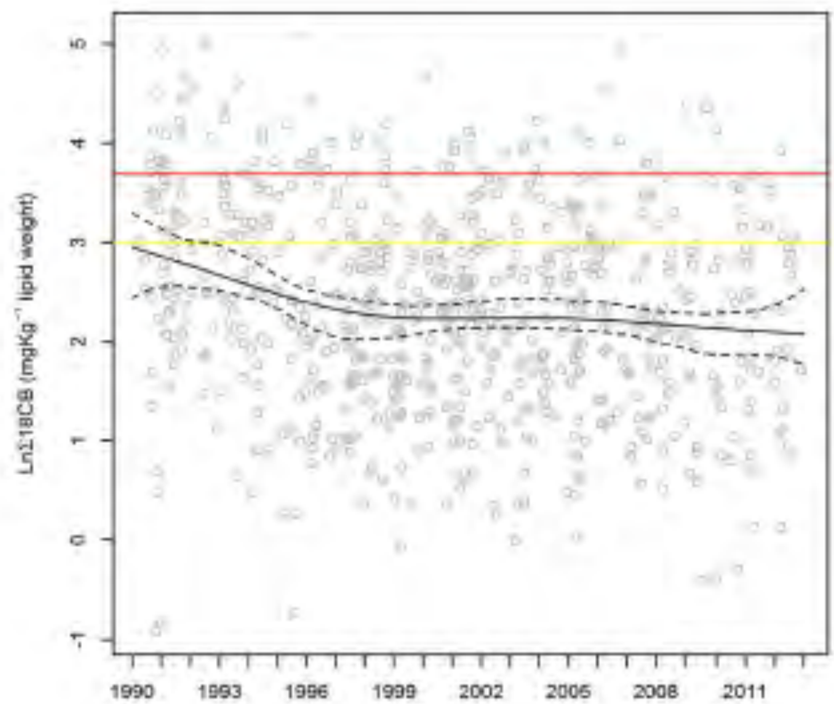


Fig. 35. Temporal trends in ΣPCBs in UK-stranded harbour porpoise (source: Jepson *et al*., 2016)

[Ln ΣPCBs (sum 18–25CB) mg/kg lipid concentrations in UK harbour porpoise blubber against date for all data for 1990–2012 (n=706). The continuous line represents the smoothed trend from a Generalized Additive Model fitted to the data. The trend is statistically significant ($p < 0.001$, $F=11.76$, residual df=701.97, trend df=3.03) against the null hypothesis of no trend. The dashed lines represent the 95% bootstrapped Confidence Intervals. The yellow line represents ln ΣPCBs equivalent to 20.0 mg/kg lipid and the red line 40 mg/kg lipid]

Although PCB lipid concentrations in UK harbour porpoises have declined since 1990, mean levels particularly in males are above the 17mg/kg ΣPCB lipid concentration thought to have adverse physiological effects (Kannan *et al*., 2000; Jepson *et al*., 2016; Fig. 35). Furthermore, a comparison of ΣPCB lipid concentrations in three other European species, bottlenose dolphin, striped dolphin, and killer whale, showed that they all had levels well above that threshold, and amongst the highest PCB contaminant burdens exhibited by those species anywhere in the world, in some groups exceeding even the very highest marine mammal PCB toxicity threshold (ΣPCB = 41 mg/kg lipid) (Jepson *et al*., 2016; Fig. 36).

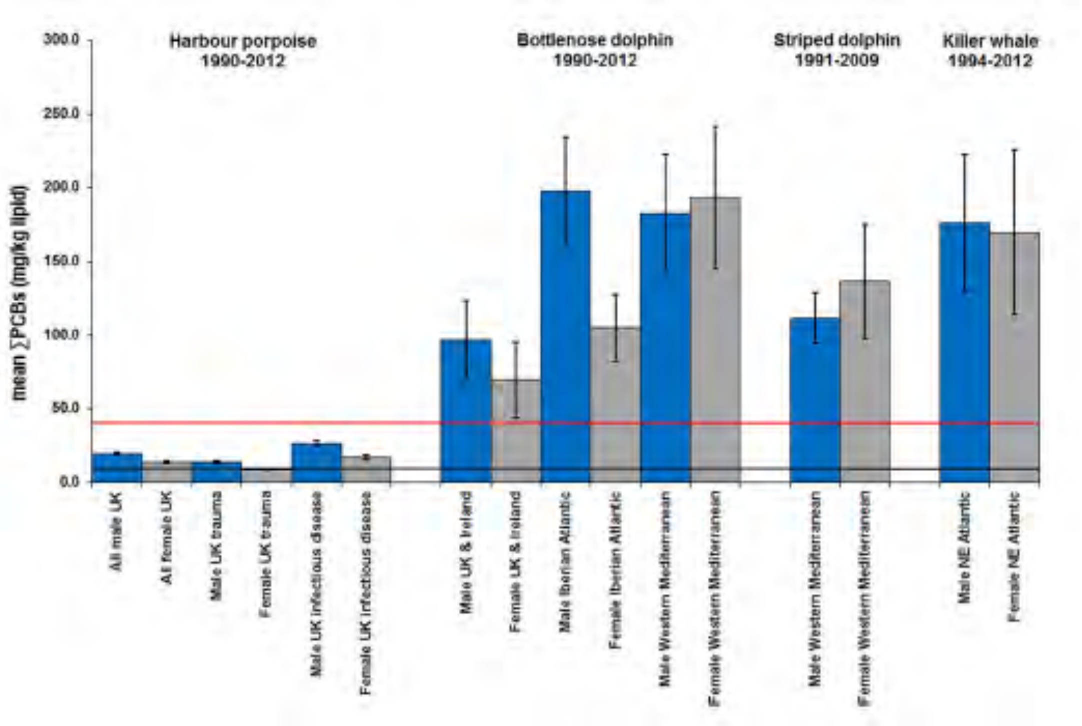


Fig. 36. Mean ΣPCBs concentrations in male and female cetaceans (four species; all ages) (source: Jepson *et al*., 2016)

[The blue bars are males and the grey bars are females. The lower black line is the equivalent ΣPCBs concentrations threshold (9.0 mg/kg lipid) for onset of physiological effects in experimental marine mammal studies. The upper red line is the equivalent ΣPCBs concentrations threshold (41.0 mg/kg lipid) for the highest PCB toxicity threshold published for marine mammals based on marked reproductive impairment in ringed seals in the Baltic Sea. Mean ΣPCBs concentrations in male (n=388) and female (n=318) UK-stranded harbour porpoises (HPs) in 1990–2012. Mean blubber ΣPCBs (mg/kg lipid) concentrations in subsets of male (n=201) and female (n=144) UK-stranded HPs that died of acute physical trauma and male (n=120) and female (n=132) HPs that died of infectious disease from the same 1990–2012 period. Mean blubber ΣPCBs (mg/kg lipid) concentrations (1990–2012) shown for stranded/biopsied male (n=29) and female (n=17) bottlenose dolphins (BNDs) from UK and Ireland; male (n = 28) and female (n=24) BNDs from Atlantic coast of Spain and Portugal and male (n=9) and female (n=11) BNDs from western Mediterranean Sea. Male (n=50) and female (n=39) striped dolphins from western Mediterranean Sea (1991–2009), and male (n=5) and female (n=19) killer whales (KW) from NE Atlantic (1994–2012). Error bars = one Standard Error of the Mean (SEM).]

Attention has been focused upon PCBs because of their widespread presence at elevated levels in the NW European marine environment, and demonstrated effects upon cetaceans. However, other POPs remain present at unacceptable levels, as indicated in Fig. 36. from OSPAR's 2010 Quality Status Report. Concentrations in the North Sea (OSPAR region II) are still widely above background values for mercury, cadmium, lead and PAHs, and above zero for PCBs, and are rated unacceptable in many, mostly coastal, areas (OSPAR, 2010). Unacceptable concentrations also persist in some urban and industrialized areas on the coasts of Regions III and IV. Overall, contamination is lowest in Region I where many of the sites monitored met the OSPAR objective of background values for heavy metals; however, concentrations of PAHs and PCBs remain widespread in the OSPAR area with more than half the sites monitored in Regions II and III (PAHs and PCBs) and IV (PCBs) at unacceptable levels. Overall, the situation is better for heavy metals, although more than 40% of sites monitored show unacceptable levels of lead in Region II and mercury in Region IV (OSPAR, 2010; Fig. 37).

In the top predators, porpoises show highest concentrations of PCBs in southern Britain and around the Irish Sea (Jepson *et al*., 2016; see Fig. 38), which are the areas where sources of this contaminant are highest. Earlier, another wide-scale study by Pierce *et al*. (2008) had found that PCB concentrations in female harbour porpoises were highest in the southern North Sea (corresponding with geographical patterns for fish). In addition, the authors found that 40% of female common dolphins, particularly those inhabiting waters off the French coast, had levels exceeding the 17 mg/kg ΣPCB lipid concentration threshold thought to have adverse physiological effects (Kannan *et al*., 2000). Murphy *et al*. (2010) found that the majority (83%) was resting mature females with high numbers of ovarian scars, suggesting that (1) due to high contaminant burdens, females may be unable to reproduce and thus continue ovulating; or (2) some females were not reproducing for some other reason, either physical or social, and therefore accumulated higher levels of contaminants.

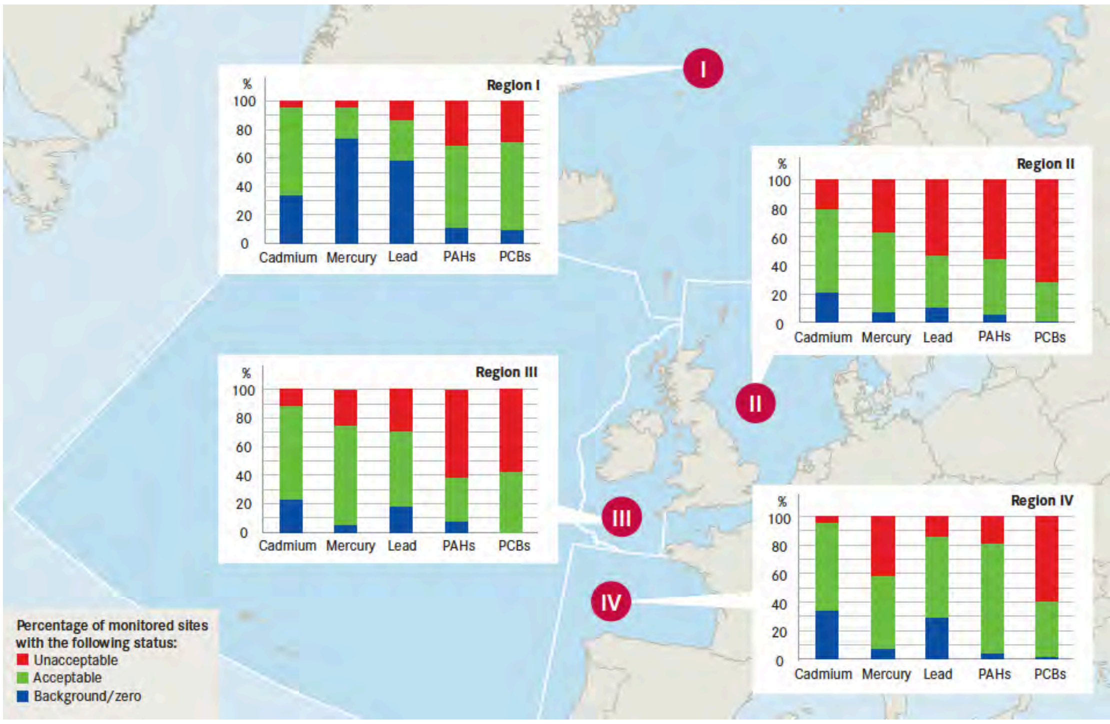


Fig. 37. Status of chemical contamination in the OSPAR area, based upon the results of the OSPAR Coordinated Environmental Monitoring Programme (Source: OSPAR, 2010)

Trace metal levels in harbour porpoise and other small cetaceans like common dolphin have been measured in a number of northern European countries (Holsbeek *et al*., 1998; Zhou *et al*., 2001; Carvalho *et al*., 2002; Das *et al*., 2004; Strand *et al*., 2005; Lahaye *et al*., 2007; Caurant *et al*., 2006). Although levels can increase with age (primarily in the liver, kidney and bone), cetaceans appear to be protected from the effects of many heavy metals by detoxification due to the presence of metallothioneins as they play a key role in essential metal homeostasis (Das *et al*., 2006). High levels of cadmium in some species tend to reflect prey preferences as this element are well known to be assimilated by cephalopods (Lahaye *et al*., 2005).

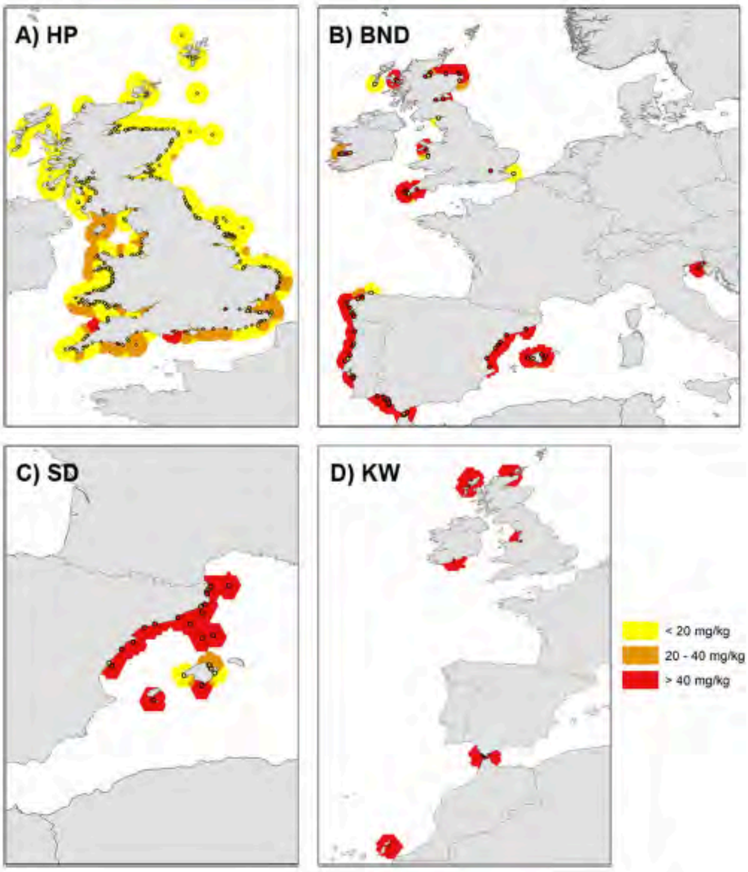


Fig. 38. Distribution map (smooth mean density kernel plots) of ∑PCBs data points in Europe – all cetacean species (all ages) from 1996-2012 (Source: Jepson *et al*., 2016)

(A) – HPs (n = 548); (B) – BNDs (n = 110); (C) – SDs (n = 71) and (D) – KWs (n = 21). Spatial distribution of ΣPCB lipid concentrations produced in Esri ArcMap 10.1 (www.esri. com). Maps are displayed in the WGS84 co-ordinate system. Data points are shown along with local averages. These averages were calculated by kernel smoothing using a polynomial order 5 kernel with power = 0, ridge parameter = 50 and bandwidth based on the spatial distribution of the observations for each species: bottlenose dolphin 0.75 degrees; harbour porpoise 0.5 degrees; killer whale 1.2 degrees; striped dolphin 0.5 degrees. Both the data points and the local averages are displayed in three colours: yellow (ΣPCB concentration = < 20 mg/ kg); orange (ΣPCB concentration = 20–40 mg/kg lw); and red (ΣPCB concentration = > 40 mg/kg lw).

3.5.5 Plastic ingestion

Marine litter, derived from both land-based and marine sources, has become an increasing concern in recent years, due to its observed impact on a wide range of marine life, particularly seabirds and sea turtles but also some cetacean species, notably beaked whales (CBD, 2012; IWC, 2013, 2014; OSPAR, 2014; Baulch and Perry, 2014). The main culprit has been plastics. The mechanism of damage has either been by entanglement in plastic sheeting, which can lead to drowning or by ingesting small plastic objects, which can lead to blockages in the stomach or intestines. Autopsies carried out on dead marine mammals and turtles have revealed that death in some cases has been linked to the ingestion of plastic waste.

A global analysis of 37 studies presenting data from before 1900 through 2011 found that the probability of plastic ingestion by sea turtles had significantly increased for leatherback turtles (*Dermochelys coriacea*) and green turtles (*Chelonia mydas*) (Schuyler *et al*., 2014). Cetaceans can also be significantly affected, and in the few observed cases from several hundred autopsies, the species affected seemed to be those that feed on cephalopods and which might have mistaken plastic bags for their prey (IWC, 2013, 2014; Baulch and Perry, 2014; OSPAR, 2014). A survey of European seas found litter in remote deep-sea areas, with the highest density in submarine canyons, and the lowest on continental shelves and ocean ridges (Pham *et al*., 2014). Plastic was the most prevalent component, with litter from fishing activities particularly common on seamounts, banks, mounds and ocean ridges.

Most plastics are extremely durable materials and persist in the marine environment for a considerable period, possibly as much as hundreds of years (OSPAR, 2014). However, plastics also deteriorate and fragment in the environment as a consequence of exposure to sunlight (photo-degradation) in addition to physical and chemical deterioration. This breakdown of larger items results in numerous tiny plastic fragments, which, when smaller than 5mm are called secondary micro plastics. Other micro plastics that can be found in the marine environment are categorized as primary micro plastics due to the fact that they are produced either for direct use, such as for industrial abrasives or cosmetics or for indirect use, such as pre-production pellets or nurdles (OSPAR, 2014).

Microplastic ingestion in cetaceans has been found in fin whale in the Mediterranean (Fossi *et al*., 2014), and in humpback whale in the southern North Sea (Besseling *et al*., 2015).

As yet, we have no evidence that ingestion of plastics is having any population level effect (Browne *et al*., 2015). Table 10 summarises data obtained from post-mortem reports on UK stranded animals examined at post-mortem between 2005-10 (Deaville and Jepson, 2011), and indicates that there has been a very low prevalence of ingestion of marine litter and also of entanglement. None of the 20 cases where evidence of plastic/litter ingestion was found resulted in any significant pathological impact on the animal and had no relationship to the cause of death (i.e. was an incidental finding). In addition, it was thought that in many cases the ingestion of marine litter may have happened in the tide line as the animal live stranded - at least 7/16 cetaceans with evidence of litter ingestion were known or diagnosed to have live stranded.

Since 2010, a few more UK-stranded cetaceans have had fragments of plastic in their stomachs. In 2011, two large pieces of plastic were found in the stomach of a Cuvier's beaked whale in Cornwall, a small plastic fragment (and a fish hook) in a minke whale stomach, and a crisp packet fragment in a harbour porpoise (Deaville, 2012) In 2012, plastic was found in the stomach of a white-beaked dolphin in Kent, a small plastic wheel and another plastic fragment in the stomach of a sei whale in Northumberland, and a plastic comb in the stomach of a northern bottlenose whale in Aberdeenshire (Deaville, 2013). In 2013, plastic was found in the stomachs of a harbour porpoise and a white-beaked dolphin in Kent, a harbour porpoise in Suffolk, and a short-beaked common dolphin in Cornwall (Deaville, 2014). In 2014, a juvenile killer whale stranded in the Western Isles, a northern bottlenose whale stranded in Highland Region, and a pygmy sperm whale stranded in North Wales all had pieces of plastic in their stomachs (Deaville, 2015). In a rare stranding of two True's beaked whales (*Mesoplodon mirus*) in Ireland, macroplastic items were identified in the stomachs of both the adults, though not in quantities likely to cause satiation and with no signs of malnutrition (Lusher *et al*., 2015). Autopsies on the German North Sea coast of a mass stranding of sperm whales in March 2016 revealed quantities of plastic including a 70 cm cover to a car engine and parts of a bucket.

Table 10. Marine litter ingestion or entanglement in cetacean, marine turtle and basking shark strandings examined at post-mortem in the UK, 2005-10 (from Deaville and Jepson, 2011)

Species	PMEs	Marine litter ingestion	Marine litter entanglement
Harbour porpoise (*Phocoena phocoena*)	459	10	0
Short-beaked common dolphin (*Delphinus delphis*)	128	3	0
Minke whale (*Balaenoptera acutorostrata*)	11	0	1
Risso's dolphin (*Grampus griseus*)	6	0	0
White beaked dolphin (*Lagenorhynchus albirostris*)	22	0	0
Bottlenose dolphin (*Tursiops truncatus*)	18	1	0
Striped dolphin (*Stenella coeruleoalba*)	22	0	0
Northern bottlenose whale (*Hyperoodon ampullatus*)	11	1	0
Atlantic white-sided dolphin (*Lagenorhynchus acutus*)	25	0	0
Long-finned pilot whale (*Globicephala melas*)	5	0	0
Sperm whale (*Physeter catodon*)	3	0	0
Humpback whale (*Megaptera novaeangliae*)	2	0	0
Sowerby's beaked whale (*Mesoplodon bidens*)	7	1	0
Loggerhead turtle (*Caretta caretta*)	18	1	0
Leatherback turtle (*Dermochelys coriacea*)	3	3	0
Kemp's ridley turtle (*Lepidochelys kempii*)	1	0	0
Basking shark (*Cetorhinus maximus*)	2	0	0
Total	**719**	**20**	**1**

NB
PMEs- post-mortem examinations
Some PME reports not available for inclusion in this table at the time of report authoring
Stomach contents not examined in; three harbour porpoises; two minke whales; two white beaked dolphins; two bottlenose dolphins; two sperm whales; one long-finned pilot whale

We conclude from studies to date that although plastic ingestion by cetaceans, particularly those feeding upon cephalopods, is of general concern, there is no evidence for population level changes in status or distribution of any of the ten species under investigation, attributable to this cause.

3.5.6 Noise Disturbance

Living in an aquatic environment where vision, touch, smell and taste have severe limitations in effective range and speed of signal transmission, cetaceans rely heavily upon sound. Different species and taxa typically utilise different frequency bandwidths of sound, which may then overlap with the sounds produced by a variety of human activities. The primary concerns are that elevated levels may cause injury, permanent threshold shifts (PTS), temporary threshold shifts (TTS), acoustic masking of communication, or behavioural disturbance (Richardson *et al*., 1995). Injury may take the form of damage to the auditory apparatus, haemorrhaging, or gas or fat emboli (Evans and Miller, 2004; Fernández *et al*., 2004, 2005; Cox *et al*., 2006). Both PTS and TTS represent actual changes in the ability of an animal to hear, usually at a particular frequency, whereby it is less sensitive at one or more frequencies as a result of exposure to sound (Finneran *et al*., 2000, 2005; Nachtigall *et al*., 2003, 2004; Cook *et al*., 2006; Southall *et al*., 2007). Masking may occur when a sound overlaps with and then 'masks' a desired signal, making the latter more difficult to detect (Clark *et al*., 2009; Erbe *et al*., 2016). Finally, behavioural responses are a demonstrable change in the activity of an animal in response to a sound, such as a change in diving behaviour, disruption of feeding or nursing, or movement away from the source (Nowacek *et al*., 2004; Tyack *et al*., 2011; DeRuiter *et al*., 2013).

Repeated exposures may affect vital activities to the extent of having population consequences (NRC, 2005). At present, we do not know whether or not short-term reactions have long-term implications on individuals or populations, and observed responses may vary among species, locations, and times of year, and depending on past exposure to seismic sounds. Baleen whales (and some toothed whales and dolphins) are long-lived compared with the majority of mammals; they mature late and have relatively low reproductive rates requiring high maternal investment in young (Evans and Stirling, 2001). Thus, the female's ability to provide adequate care to her offspring during a prolonged period of dependency is critical to the continued recovery and long-term viability of these populations and supports the need to avoid disturbance in certain seasons or locations.

In the context of this study, four main sources of anthropogenic sound will be considered: shipping, seismic, marine construction, and active sonar.

Shipping In the mid-19th century, a new source of sound started to fill the ocean, driven by the rapid spread of mechanical propulsion in the shipping industry. Shipping has long been recognised as an important anthropogenic sound source (Wenz, 1962).

The global commercial shipping fleet expanded from about 30,000 vessels (of about 85 million gross metric tons) in 1950 to more than 85,000 vessels (about 525 million gross metric tons) in 1998 (NRC, 2003). About 90 percent of world trade (in gross tonnage) depends on ship transport and, apart from declines during global economic downturns, the gross tonnage of goods transported by sea has steadily increased since the early 1970s.

Large vessels typically have sound source levels of 160-220 dB re 1μPa @ 1 m over a bandwidth of 5-100 Hz, with peak energy around 25 Hz (Richardson *et al.*, 1995; NRC, 2003).

The volume of cargo transported by sea has been doubling approximately every twenty years (http://www.marisec.org/shippingfacts/ worldtrade/volume-world-trade-sea.php), resulting in an increase in anthropogenic sound from this source. Although the measurement of sound in relation to these changes has been mostly local and is incomplete, the current estimate is that increased shipping has been accompanied by a significant increase in anthropogenic sound at frequencies below 500 Hz. From 1950 to 2000, the shipping contribution to ambient sound at some locations increased by as much as 15 dB, corresponding to an average rate of increase of approximately 3 dB per decade (Andrew et al., 2002, 2011; Hildebrand, 2009; Chapman and Price, 2011; Frisk, 2012). Shipping is probably the greatest single source of human-generated sound in the ocean (Tyack *et al*., 2015).

The importance of shipping sound for marine life is still largely unknown. As noted above, shipping sound has the potential to mask the communication signals of marine mammals (and fish), and both taxa have been shown to change behavior in reaction to these sounds (Tyack, 2008). However, even though predictions based on theory indicate that communication ranges can be decreased as a result of increased sound levels, many species may have developed mechanisms to compensate for masking, for example, increasing the source level of their sounds when located in an increased noise environment (Parks *et al*., 2010). There are also large differences in potential effects between deep and shallow waters and among the taxonomic groups affected (Tyack *et al*, 2015).

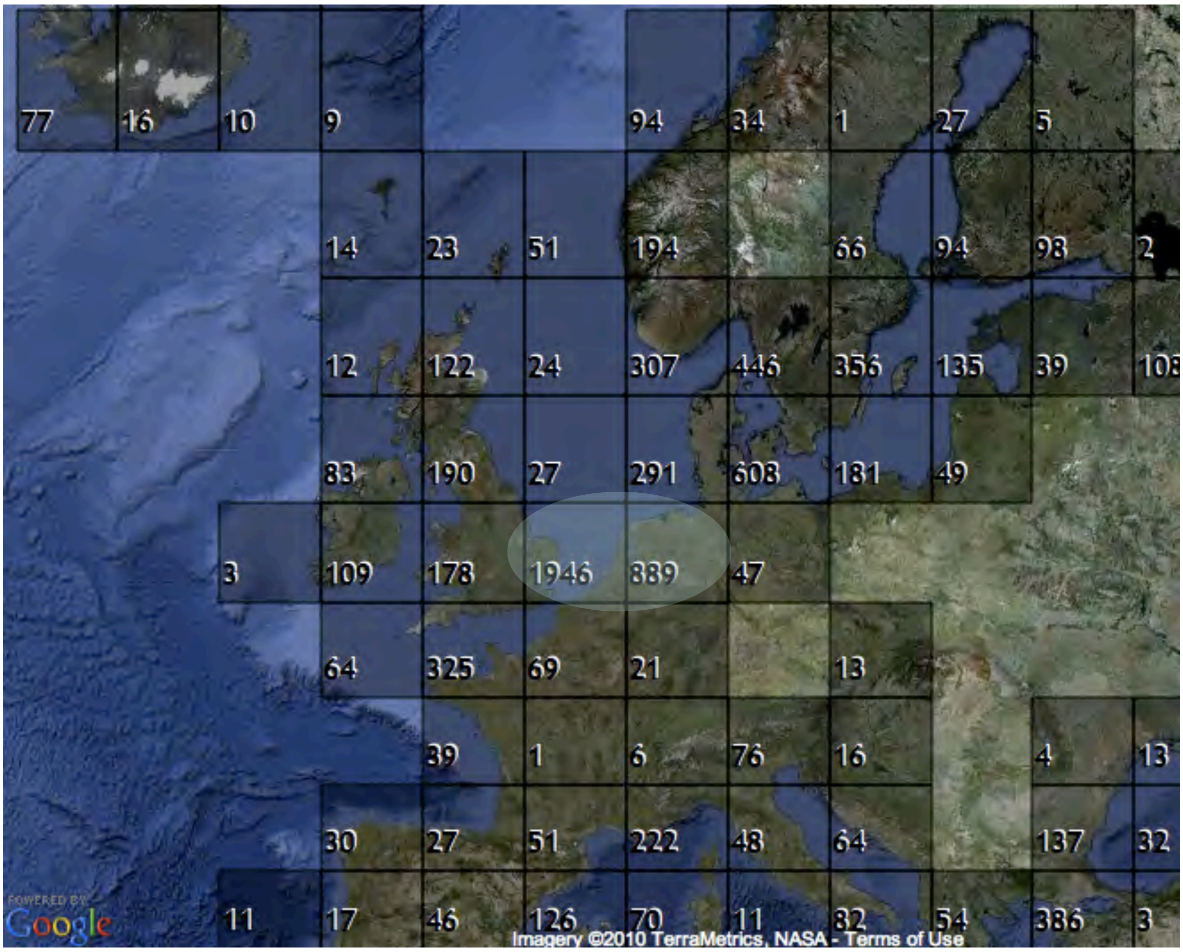


Fig. 39. Number of ships recorded by AIS in NW European seas on a sample day - 15 Aug 2010 (oval area highlighted is shown at higher resolution in Fig. 40) (Source: Evans *et al*., 2011)

Soundscape measurements exist for only a limited number of relatively small areas. However, shipping tends to be concentrated within long-standing shipping lanes (see Fig. xx). Thus it is possible to identify those areas in NW Europe where noise from vessels is likely to be greatest. This strongly highlights the southern North Sea, Strait of Dover and eastern part of the English Channel (Evans, 2007; Evans, *et al*., 2011; Figs. 39 and 40).

That region has long had low biodiversity and densities of cetaceans (Evans, 1980, 2008; Evans *et al*., 2003; Reid *et al*., 2003). However, since the mid 1990s, harbour porpoises in particular have increased substantially in this area, thought to be in response to an increase in food resources there (Evans, 1990; Evans *et al*., 2003; Kiszka *et al*., 2004, 2007; Haelters and Camphuysen, 2009; Hammond *et al*., 2013). Thus if background noise levels are having a negative effect upon cetaceans, it is not sufficient to prevent the recent increase in porpoise numbers in the noisiest sector.

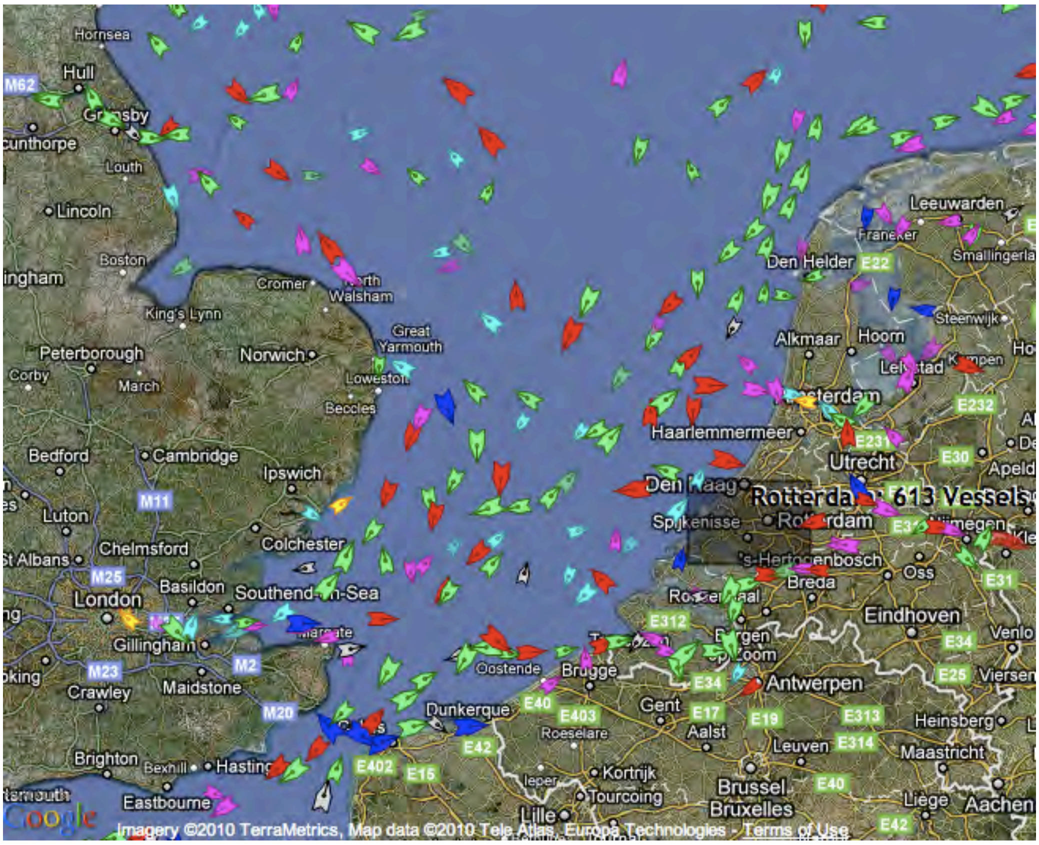


Fig. 40. Distribution of Shipping in the Southern North Sea and English Channel, 15 Aug 2010 (from Evans *et al*., 2011)

Seismic The second most pervasive source of noise from human activities is that of seismic, produced during geophysical exploration for oil and gas. Since the present study relates specifically to possible impacts of seismic on cetaceans, this will be reviewed in more detail than for other human activities.

Seismic surveys produce short duration broadband impulse sounds with peak source levels of c. 220-255 dB re 1μPa peak at 1 m (Richardson *et al*., 1995; Evans and Nice, 1996; Nowacek *et al*., 2007). The sound is directed downwards towards the sea floor with most energy <300 Hz although high frequencies up to at least 15 kHz may also be produced (Good and Fish, 1998; Madsen *et al*., 2006). Since peak frequencies for seismic sound directly overlap the vocalisations and estimated hearing range of baleen whales, these are considered to be more likely to be disturbed than odontocetes (Richardson *et al*., 1995; Evans and Nice, 1995; Nowacek *et al*., 2007). Several studies have shown negative reactions from baleen whales (mainly bowhead whales, gray whales and humpbacks), including deflections from migratory routes, avoidance behaviour, cessation of feeding, and changes in both aerial and surfacing behaviour (Reeves *et al*., 1984; Richardson *et al*., 1986; Würsig and Greene, 1986; Ljungblad *et al*., 1988; McCauley *et al*., 2000; Miller *et al.*, 2005; Stone and Tasker, 2006; Gailey *et al.* 2007; Yazvenko *et al.*, 2007a,b; Abgrall *et al*., 2008; OSPAR, 2009).

Mysticetes The responses of bowhead whales to seismic have been particularly studied, and show great variability depending on the activity of the whales – whether they are migrating or feeding. On migration, they are particularly responsive, with substantial avoidance occurring out to distances of 20–30 km from a medium-sized airgun source, where received sound levels were in the order of 130 dB re 1 μPa_{rms} (Miller *et al*., 1999; Richardson *et al*., 1999). During the summer feeding season, however, bowheads do not appear to be as sensitive to seismic sources, bowheads typically beginning to show avoidance reactions at a received level of about 160–170 dB re 1 μPa_{rms} (Richardson *et al*., 1986; Ljungblad *et al*., 1988; Miller *et al*., 2005a,b; Harris *et al*., 2007). Nevertheless, there was statistical evidence of subtle changes in surfacing, respiration and diving cycles when feeding bowheads were exposed to lower-level pulses from distant seismic operations (see Richardson *et al.,* 1986). This suggests that feeding whales do respond to seismic sounds, but the need to feed apparently generally overrides the tendency to move away.

The responses of western Pacific gray whales to 3D seismic exploration have been studied recently on their feeding grounds off Russia's Sakhalin Island. Although no measurable effect on bottom feeding activity was detected (Yazvenko *et al*., 2007a), and nor did overall numbers change, some gray whales redistributed themselves within the feeding area when the seismic survey was fully operational, and there were changes in movement patterns (Yazvenko *et al*., 2007b; Gailey *et al*., 2007).

Studies of the responses of migrating humpbacks to seismic sound (both full-scale seismic surveys and experimental exposures to a single airgun) in Western Australia showed a gender difference. Male humpback whales were relatively tolerant of seismic sound, some even approaching the vessel, whereas females, by contrast, showed strong avoidance behaviour at a range of 5-8 km from the full-scale array, and maintained a stand-off range of 3-4 km (McCauley *et al*., 1998, 1999, 2000a,b). Typical received levels at 5 km were measured as 162 dB re 1 $\mu Pa_{peak\text{-}peak}$. More recently, a large-scale and carefully designed experimental behavioural response study (called BRAHSS) of migrating humpbacks along the east coast of Australia (Cato *et al*., 2013) found that whale groups responded by decreasing both dive time sand speed of southwards movement although the magnitude of the response was not related to the proximity of the source level, the received level of the airgun, the tow path direction, or the exposure time (Dunlop *et al*., 2015). There was no evidence of orientation of the groups towards, or away from, the source vessel during seismic sound generation (Dunlop *et al*., 2015).

Some evidence of displacement was also found collectively amongst balaenopterid whale species (blue, sei, fin, and minke whales) in UK waters, the median distances for closest point of approach being significantly larger (~1600 m vs. 1000 m) and whales tending to head away from the vessel during seismic vs. non-seismic periods, although overall sighting rates did not differ (Stone and Tasker, 2006). In eastern Canada (The Gully MPA), Moulton and Miller (2005) also found little difference in sighting rates and initial average sighting distances of balaenopterid whales when airguns were operating (mean = 1324 m) vs. silent (mean = 1303 m) but there were indications that these whales were more likely to be moving away when seen during airgun operations. Baleen whales at the average sighting distance during airgun operations would have been exposed to sound levels (via direct path) of about 169 dB re 1 μPa_{rms} (Moulton and Miller, 2005). Elsewhere off Newfoundland, Moulton *et al*. (2005, 2006a,b) found variable responses between years, with some individuals showing localized avoidance of seismic operations.

It should be noted that avoidance distances often exceed the distances at which boat-based observers can see whales, so observations from the source vessel may be biased. Studies indicate that monitoring over broader areas may be needed to determine the range of potential effects of some larger seismic surveys (Miller *et al.*, 1999; Bain and Williams, 2006; Moore and Angliss, 2006).

Studies of gray, bowhead, and humpback whales have determined that received levels of pulses in the 160–170 dB re 1 μPa_{rms} range seem to cause obvious avoidance behaviour in a substantial fraction of the animals exposed. In the case of migrating bowhead whales, avoidance extends to lower received sound levels and larger distances. However, in other situations, various mysticetes tolerate exposure to full-scale airgun arrays operating at closer distances, with only localized avoidance and minor changes in activities.

Odontocetes The responses of toothed whales and dolphins to seismic are less well known, with no studies similar in size and scope to those of humpback, bowhead, and gray whales. There have been a few systematic studies on sperm whales (Jochens *et al.*, 2006; Miller *et al.*, 2006; Miller *et al.*, 2009), and there is an increasing amount of information about responses of various odontocetes to seismic surveys based on monitoring studies (e.g., Goold, 1996; Smultea *et al.*, 2004; Moulton and Miller, 2005; Bain and Williams, 2006; Holst *et al.*, 2006; Stone and Tasker, 2006).

Seismic operators and marine mammal observers regularly see dolphins and other small odontocetes near operating airgun arrays, but in general there seems to be a tendency for most delphinids to show some limited avoidance of operating seismic vessels, in the order of 1 km or less (e.g., Moulton and Miller, 2005; Holst *et al.*, 2006). Some dolphins (and Dall's porpoises) seem to be attracted to the seismic vessel and floats, and some ride the bow wave of the seismic vessel even when large arrays of airguns are active (e.g., MacLean and Koski, 2005; Moulton and Miller, 2005). Nonetheless, small toothed whales often tend to head away, or to maintain a somewhat greater distance from the vessel, when a large array of airguns is operating than when it is silent (e.g. Smultea *et al.*, 2004; Holst *et al.*, 2005a, 2006; Stone and Tasker, 2006), and some species (e.g. beluga whale) even show long-distance avoidance of seismic vessels, in the order of 10-20 km (Miller *et al.*, 2005a; Harris *et al.*, 2007).

An analysis of observations taken during 201 seismic surveys in UK and adjacent waters indicated that small odontocetes showed a greater range of responses to seismic surveys than did mysticetes or larger odontocetes, including significant declines in sighting rates during periods of seismic surveys (Stone and Tasker, 2006). On the other hand, larger odontocetes (long-finned pilot whales, killer whales and sperm whales) showed little response to airgun activities and no reduction in sighting rates during periods of seismic surveys.

Variable results were also found during two seismic surveys off Newfoundland and Labrador, in 2004 and 2005 (Moulton *et al.*, 2005, 2006a). During both surveys, dolphin sighting rates (taking temporal variation into consideration) were higher during non-seismic periods than during seismic periods, although this difference was only statistically significant in 2004. The mean closest point of approach of dolphins was significantly closer during non-seismic periods (652 m) vs. seismic periods (807 m) in 2005, but the difference was not statistically significant in 2004 (705 m vs. 665 m, respectively). On the other hand, there were no significant differences in the sighting rates of closest points of approach in large odontocetes, primarily sperm whales. Observations from the seismic vessel indicated that some odontocetes exhibited localized avoidance of seismic operations (Moulton *et al.*, 2005, 2006a).

There are no specific data on the behavioural reactions of beaked whales to seismic surveys, although they tend to avoid approaching vessels of other types (Sorensen *et al*., 1984; Würsig *et al*., 1998). Recent studies show little evidence of reactions by sperm whales to airgun pulses, contrary to earlier indications (Jochens *et al*., 2006).

Odontocete reactions to large arrays of airguns are variable. For delphinids, significant disturbance seems to be confined to a smaller radius than has been observed for the more responsive of the mysticetes, although strong reactions seem to be largely limited to the area within 1 or 2 km of airgun arrays. Given the available data on typical received levels of airgun array sound at those distances, a 170 dB re 1 μPa_{rms} disturbance criterion may be more appropriate for delphinids than the ≥160 dB criterion that is often used (Abgrall *et al*., 2008).

Results from sperm whale tagging studies in the northern Gulf of Mexico have shown that neither gross diving behaviour nor direction of movement changed for any of the eight tagged individuals exposed to the onset of seismic airgun sounds (a gradual ramp-up at ranges of 7–13 km or during full-power exposures 1.5–12.8 km from the airguns) (Jochens *et al*., 2006). However, some changes in foraging behaviour were observed that suggested avoidance of deep dives near operating airguns. Foraging behaviour was disrupted by airguns at exposure levels ranging from <130 to 162 dB re 1 μPa (peak-peak) at distances of roughly 1–12 km from the sound source. These results raise questions about the efficacy of ramp-up as a mitigation strategy for sperm whales given that they did not swim away from oncoming seismic vessels, although it is possible that in this region they were already habituated to seismic activities (Jochens *et al*., 2006). Miller *et al.* (2009) later found a reduction in swimming effort accompanied by an apparent reduction in foraging when sperm whales were exposed to seismic surveys.

Shore-based observations of western gray whales on their feeding grounds indicated that most measures of surfacing, respiration and diving behaviour showed no significant correlation with seismic survey variables (Gailey *et al*., 2007). At higher received sound energy exposure levels, whales were observed to actually stay under water longer between respirations (contrary to some earlier findings from gray and bowhead whales).

Some baleen and toothed whales are known to continue calling in the presence of seismic operations and often can be heard between the seismic pulses (e.g., Richardson *et al*., 1986; McDonald *et al*., 1995; Greene *et al*., 1999a,b; Nieukirk *et al*., 2004; Smultea *et al*., 2004). Although there was a report that sperm whales ceased calling when exposed to pulses from a very distant seismic ship (Bowles *et al*., 1994), a more recent study found that sperm whales off northern Norway continued calling in the presence of seismic pulses (Madsen *et al*., 2002), and the same was found in the Gulf of Mexico (Tyack *et al*., 2003; Smultea *et al*., 2004; Jochens *et al*., 2006). Dolphins and porpoises commonly are heard calling while airguns are operating (e.g., Gordon *et al*., 2004; Smultea *et al*., 2004; Holst *et al*., 2005a,b), although Goold (1996) found a reduction in vocalizing common dolphins during periods of seismic *vs* non-seismic sound.

An early study in the Canadian Beaufort Sea showed that bowhead whales continue to call in the presence of airgun sounds, with the types of calls being unchanged (Richardson *et al*., 1986). However, a subsequent study in the Alaskan Beaufort Sea found evidence of a reduction in bowhead calls when airgun sounds were present (Greene *et al*., 1999a,b). A problem in interpreting some of these findings, however, is the difficulty in distinguishing between reduction in call rates and actual movement out of the area, both of which may occur (Miller *et al*., 1999; Richardson *et al*., 1999).

There have been recent attempts to differentiate between the two using passive acoustics for bowhead and beluga whales, but with limited success (Funk *et al*., 2007).

As noted earlier, the longer-term consequences of seismic are difficult to determine and for that reason there is scant information available. A recent study in the Moray Firth, Northeast Scotland, using passive acoustic monitoring and digital aerial surveys found group responses amongst harbour porpoises to airgun noise from a two-dimensional seismic survey over ranges of 5–10 km, at received peak-to-peak sound pressure levels of 165–172 dB re 1 μPa and sound exposure levels (SELs) of 145–151 dB re 1μPa^2/s. However, animals were typically detected again at affected sites within a few hours, and the level of response declined through the 10-day survey. Overall, acoustic detections decreased significantly during the survey period in the impact area compared with a control area, but this effect was small in relation to natural variation. This area has few baleen whales and so it is not possible to infer longer-term effects upon mysticetes.

Marine Construction: explosions, dredging and drilling Marine construction activities, particularly explosions, have potential to cause physical damage as well as behavioural disturbance (Ketten, 1995; Nowacek *et al*., 2007; OSPAR, 2009; Anderwald *et al*., 2013; Culloch *et al*., 2016).

Underwater explosions are used not only in construction but also to remove unwanted subsea structures including military ordinance. They are one of the strongest point sources of any anthropogenic sound, starting with an initial shock pulse followed by a succession of oscillating bubble pulses (Richardson *et al*., 1995; OSPAR, 2009). Source levels vary with the type and amount of explosives used, the water depth at which the explosion occurs and can range from 272 to 287 dB re 1 μPa zero to peak at 1 m distance (1-100 lb. TNT). Frequencies are rather low (range 2 - ~1 kHz; main energy between 6–21 Hz; duration <1-10 ms; Richardson *et al*., 1995; NRC, 2003). The disposal of unexploded military ordnance dumped during Word War II in coastal areas of the southern North Sea has recently caused concern over potential impacts on marine mammals (Koschinski, 2011; Benda-Beckmann *et al*., 2015; Koschinski and Kock, 2015).

There have been few studies of the impacts of explosions upon cetaceans. Todd *et al.* (1996) did not find any changes in behaviour of humpback whales to blasts during the development of an offshore oil platform (received SPL = 140-153 dB re 1 μPa rms at 1.8 km). Madsen and Møhl (2000) found no acoustic reactions of five sperm whales to distant detonators at received sound pressure levels of 179 dB re 1 μPa rms, which they postulated may have been because the detonator noise resembled sperm whale clicks and might therefore have been perceived as signals from conspecifics. There were some other issues with this study that makes it difficult to interpret the findings (Nowacek *et al*., 2007).

Finneran *et al.* (2000) exposed two trained dolphins and one beluga to sounds resembling distant blast explosions. They observed no auditory shift (i.e. TTS) greater than 6 dB to levels up to 221 dB re 1 μPa peak-peak. Behavioural changes in the form of delayed approaches to the test station and 'avoiding' the start station, were observed at 196 and 209 dB re 1 μPa peak-peak for the two dolphins, respectively, and at 220 dB re 1 μPa peak-peak for the beluga, and these alterations continued at higher levels. However, a number of caveats to this study included the fact that the signals they used were for very distant ones and signals from nearby explosions would differ in both level and structure; and they measured masked-hearing thresholds. In the absence of masking noise, larger threshold shifts might have been measured, whilst the fact that the animals were trained with rewards for tolerating high levels of noise meant that behavioural disruption would likely be observed at lower levels in other contexts (Nowacek *et al*., 2007).

Given the comparatively low source levels, injuries from either dredging or drilling operations are unlikely in marine mammals, except very close to the source (see Southall *et al*. (2007) for suggested noise exposure criteria). There is no documented case of injury caused by pile driving in the wild, but this should be interpreted with caution since studies are very limited and observations of injury are almost impossible to obtain under natural conditions. Temporary threshold shift (TTS) caused by signals resembling explosives has been investigated in captive bottlenose dolphins and beluga whales. No TTS was observed to levels up to 221 dB re 1 μPa peak to peak (Finneran *et al*., 2000). Again, as this is the only study looking at TTS induced by sounds resembling construction noises, general conclusions are limited. Richardson *et al.* (1995) report some rather poorly documented cases of injury and death of marine mammals thought to have been caused by explosions. Ketten (1993) reports injury in the ears of two humpback whales stranded after underwater explosions.

Todd *et al*. (1996) studied humpback whale movements and behaviour in relation to blasts occurring during the development of a concrete oil production platform in Trinity Bay, Newfoundland. RLs of sound from Tovex charges varied with the size of charge detonated but were measured, at a distance of 1.8 km, to be 140–153 dB re 1 μPa rms with the peak amplitude occurring at ~400 Hz. The authors stated that the techniques used to document potential response(s) were designed to provide equal sampling of areas less than and greater than 10 km from the source, but the effort was not otherwise quantified as a function of range from the source. Short-term behavioural parameters of whales were apparently collected on an *ad lib* basis, and the authors reported no detectable changes in respiration rates nor occurrence of abrupt surface reactions around the time of blasts, although *ab lib* sampling would not necessarily produce data sufficient to detect changes. There were no reported differences in re-sighting rates or distance to the site of the explosions between blasting and non-blasting periods during the observation period. Most of the whale sightings during this study occurred between 3 and 9 km from the blast site, and individual animals within 10 km of the blast site were sighted significantly more often than animals in greater range categories: 10–20 and 20+ km. Residency times (the mean number of days a whale was re-sighted) were longest closest to the blast site, though again the effort with various range categories was not reported. The authors suggested that the lack of response to the blasts could be attributed to high prey abundance, and while they mention field observations showed abundances of prey throughout and outside the study area, no quantitative estimates were provided. Also, to account for the lack of response, they suggested that the whales may have habituated to the blasts since the study occurred midway through the blast schedule. Associated with this same construction activity, Borggaard *et al.* (1999) studied humpback whale abundance, distribution and movements in relation to blasting, dredging and vessel traffic in Trinity Bay over a longer period (1992–95), although the authors did not report any information about the levels or characteristics of sound received by the whales.

In July 2011, 70 long-finned pilot whales swam into the Kyle of Durness, a shallow tidal inlet east of Cape Wrath, Europe's largest live bombing range. Despite attempts to herd them back out to sea, 39 were left stranded by the tide, and 19 ultimately died. Three 1,000 lb bombs were detonated in the sea 24 hours before the mass stranding, and were concluded to be the most likely cause of the mass stranding

(Brownlow *et al*., 2015).

Richardson *et al*. (1995) reviewed the observed effects upon marine mammals of stationary drilling activities (and simulated drilling), and found avoidance reactions in a number of species (particularly baleen whales) but only when RL exceeded around 120 dB re. 1 μPa,, and on either initial exposure or an increase in sound levels.

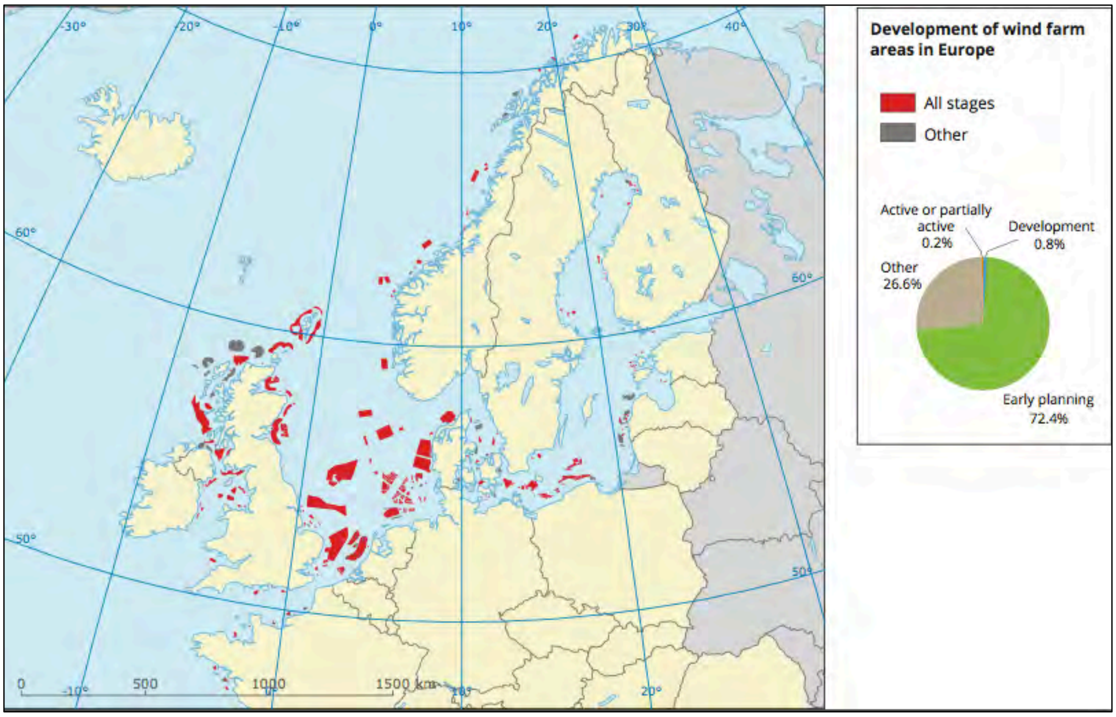


Fig. 41. Map of wind farm areas in all stages of development in Europe
The pie chart shows the different levels of development as percentages
(source: European Environment Agency, 2005: http://www.eea.europa.eu/data-and-maps/figures/development-of-wind-farm-areas)

In recent years there have been many studies on the effects upon marine mammals of wind farm construction and operation. A number of reviews have been published (Teilmann *et al*., 2006; Madsen *et al*., 2006a; Evans, 2008; Thomsen, 2010; Mann and Teilmann, 2013). Development proposals for offshore wind farms have generally considered large mono-pile designs with diameters of between 4 and 6 metres, and likely to increase. This size of driven pile has the potential to give rise to peak-to-peak source levels in excess of 250 dB re 1 μPa @ 1 m (Nedwell *et al*., 2008). Shallow areas and pile driving are often preferred for economic reasons, so that the cetacean species most often likely to coincide at construction sites is the harbour porpoise.

Effects largely are negative and relate to reduced presence and/or echolocation activity of harbour porpoise in the neighbourhood of the area exposed to pile driving, with some effects (e.g. at Nysted, Danish Baltic) occurring out to beyond 20 km from the noise source, and lasting more than ten years, although in other cases (e.g. OWEZ, The Netherlands) there was no apparent negative effect (Tougaard *et al*., 2008a, b; Carstensen *et al*., 2008; Tougaard *et al*., 2009; Scheidat *et al*, 2010; Brandt *et al*., 2011; Teilmann and Carstensen, 2012). The differences may relate to the context in which the species uses the affected area and confounding factors (e.g. increased prey abundance for other reasons). Koschinski *et al.* (2003) observed reactions of harbour porpoises and harbour seals to the broadcast of wind generator noise, and reported source levels but did not report any information on RL.

Wind farm developments started in Europe in the early 1990s, with the construction of Horns Rev in the Danish North Sea, followed shortly after by one at Nysted in the Danish Baltic, and others (e.g. Kriegers Flak) since then. In UK waters, projects were proposed in three licensing rounds (Round 1 in 2001, involving 18 sites in England and Wales; Round 2 in 2003, in three areas: Greater Wash, Greater Thames and Irish Sea; and Round 3 in 2010 featuring nine zones, with construction from 2014 onwards). In addition, sites in Scottish and Northern Irish waters have been investigated since 2012. In Dutch waters, six wind farms have been constructed or proposed, since the 1990s. The first large one was the OWEZ demonstration wind farm near Egmond aan Zee started in 2003, with operation from 2007, but other large ones exist (Eneco Luchterduinen and Princess Amalia), or are underway (Gemini). Germany entered the wind farm arena rather later but now has six under construction or in operation. The distribution of wind farm areas at all stages of development is given in Fig. 41.

Active Sonar Active sonar, operating with sound source levels of up to 245 dB re 1µPa @ 1 m at frequencies mainly between 1 and 150 kHz, is frequently used for fish-finding, oceanography, charting and in military activities (for example locating submarines). Since the mid-1990s, concern has been expressed over the potential impact these sounds may have upon cetaceans (particularly deep diving toothed whales of the Sub Order Odontoceti such as the beaked whales, family Ziphiidae), and post mortem studies of mass stranded animals in the Bahamas, Madeira and the Canaries have revealed multifocal haemorrhaging and ear damage (Evans and Miller, 2004; Cox *et al.*, 2006). Naval sonars are high intensity sound sources that operate within the frequency ranges that marine mammals can hear. Southall *et al.* (2007) proposed an acoustic injury threshold to cetaceans for sounds such as sonar with multiple pulses, of 198 dB re 1 µPa/s in terms of SEL. Most military sonars have source levels in the 220-240 dB re 1 µPa at 1 m range. This means that for one 1s pulse of a sonar to pose a risk to a cetacean, it would have to be within 10-100 m of the sonar. Naval ships move rapidly enough that it would be unlikely for multiple pulses to add enough energy to increase SEL over the threshold at greater ranges, although it should be noted that the SEL criterion may not accurately predict TTS particularly if exposures are of long duration (Tyack, 2015).

Atypical mass strandings of whales especially beaked whales variously linked to naval exercises have been well documented (Frantzis, 1998; Evans and England, 2001; Evans and Miller, 2004; Cox *et al.,* 2006; D'Amico *et al.*, 2009). Veterinary pathologists analysing whales from several of these strandings also identified decompression symptoms, suggesting that whales do not just die from stranding but may be injured or die at sea (Fernández *et al.*, 2004, 2005; Jepson *et al.*, 2005b). The possibility of a behavioural response to naval sonars has been investigated in a series of controlled exposure experiments (Tyack *et al.*, 2011; DeRuiter *et al.*, 2013). Beaked whales were found to respond to mid-frequency active sonar (MFA) playback at levels of 98, 127 (DeRuiter *et al.*, 2013), and 138 dB re 1 µPa (Tyack *et al.*, 2011). Responses were unusually slow ascent rates, unusually long inter-deep-dive intervals, and a premature cessation of echolocation used for foraging. They also included strong and prolonged directional movement away from the sound source. Although beaked whales appear to be particularly susceptible, blue whales have also been recorded responding to mid-frequency naval sonar (Goldbogen *et al.*, 2013).

A number of areas in NW Europe have traditionally been used for military exercises, with mid-frequency sonar applied increasingly since the 1960s. Around the British

Isles, where Joint Warrior exercises are practiced, these include 1) offshore in the NW Approaches to Scotland (area bounded by 56°00'N-56°30'N, 07°00'W-08°00'W); 2) coastal waters in the Sea of Hebrides (areas: Hebrides South, Hebrides Central, Hebrides North, Canna, Hawes, and Neist); 3) North Minch (areas: Trodday, Shiant, North Mivh, South Minch, Tiumpan, Stoer, Ewe); 4) the central west English Channel (area bounded by 58°30'N-59°00'N, 05°00'N-06°00'W); and 5) offshore in the westernmost English Channel (area bounded by 58°00'N-59°00'N, 08°00'N-10°00'W) (see Fig. 42).

a) Scotland West Coast Areas

b) South Coast UK

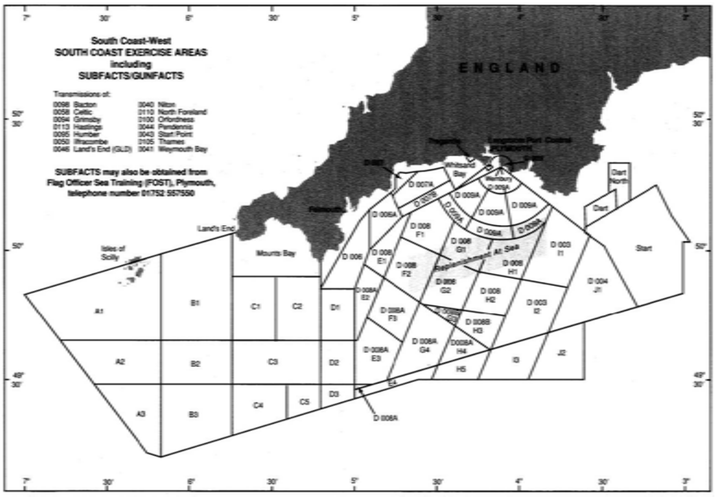


Fig. 42. Joint Warrior Training Exercise Areas
(maps provided courtesy of the Royal Navy)

In Scotland, within Areas 1-3, sonar activity typically lasts for 6 hours each day for up to three ships and helicopters. Within Area 4, sonar activity lasts for 6 hours each day for a 3-day period in the second week of the exercise. Area 5 is used for 3—4 days approximately every third exercise for passive ASW and Sonar 2087 Active operations (sonar 2087 operates at frequencies of 800 Hz – 1.6 kHz). These exercises take place twice a year, once in the spring and once in the autumn. In Southern England, the areas are typically used for 43 weeks of the year and for 22 of those weeks, they may have three or more sonar fitted ships taking part (Royal Navy, *pers. comm.*).

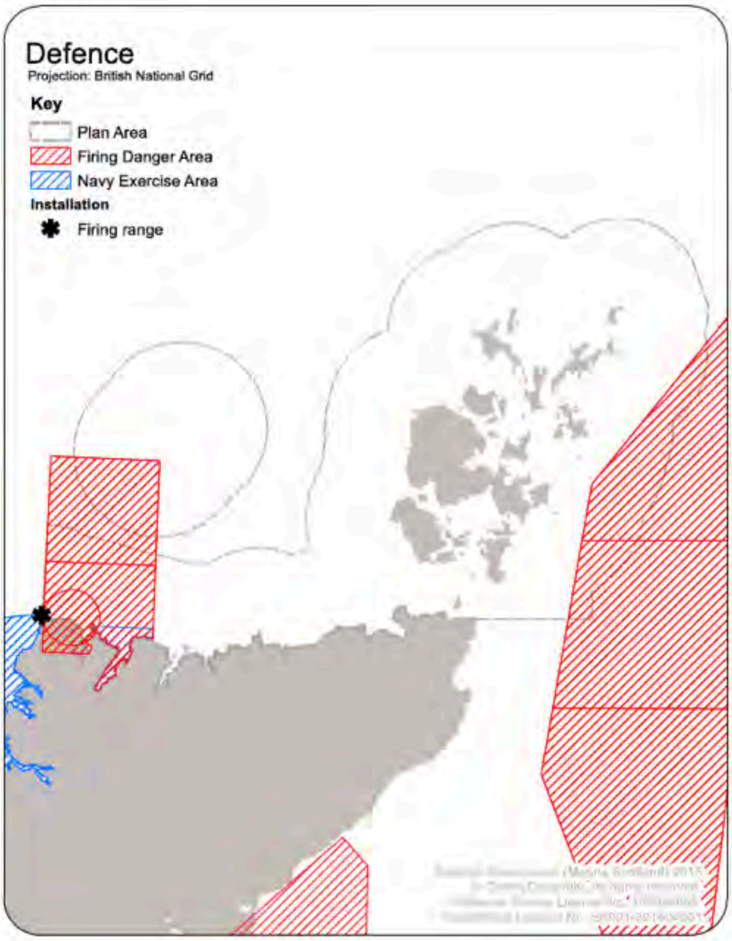


Fig. 43. Defence Locations in the Pentland Firth/Orkney area and adjacent waters of the Moray Firth (Source: Marine Scotland: http://www.gov.scot/Publications/2015/06/9524/10)

Elsewhere, The Ministry of Defence (MoD) uses a number of areas in or adjacent to the Pentland Firth and Orkney Waters, mainly for training purposes. In particular, Cape Wrath is an exercise area, firing range and a firing danger area, along with the Navy exercise area immediately west, which extends right down the west coast (referred to above). There is also an exercise area and a firing danger area, which covers sections of the Moray Firth and runs parallel offshore up to the northeast tip of the Orkney Islands. Figure 43 summarises the military activity in the Pentland Firth/Orkney region and adjacent waters. The Vulcan Naval Reactor Test Establishment, operated by the MoD is located next to the Dounreay site in Caithness.

The Firth of Forth is also used for some Naval exercises. A map illustrating locations of all defence activities in Scotland is given in Figure 44.

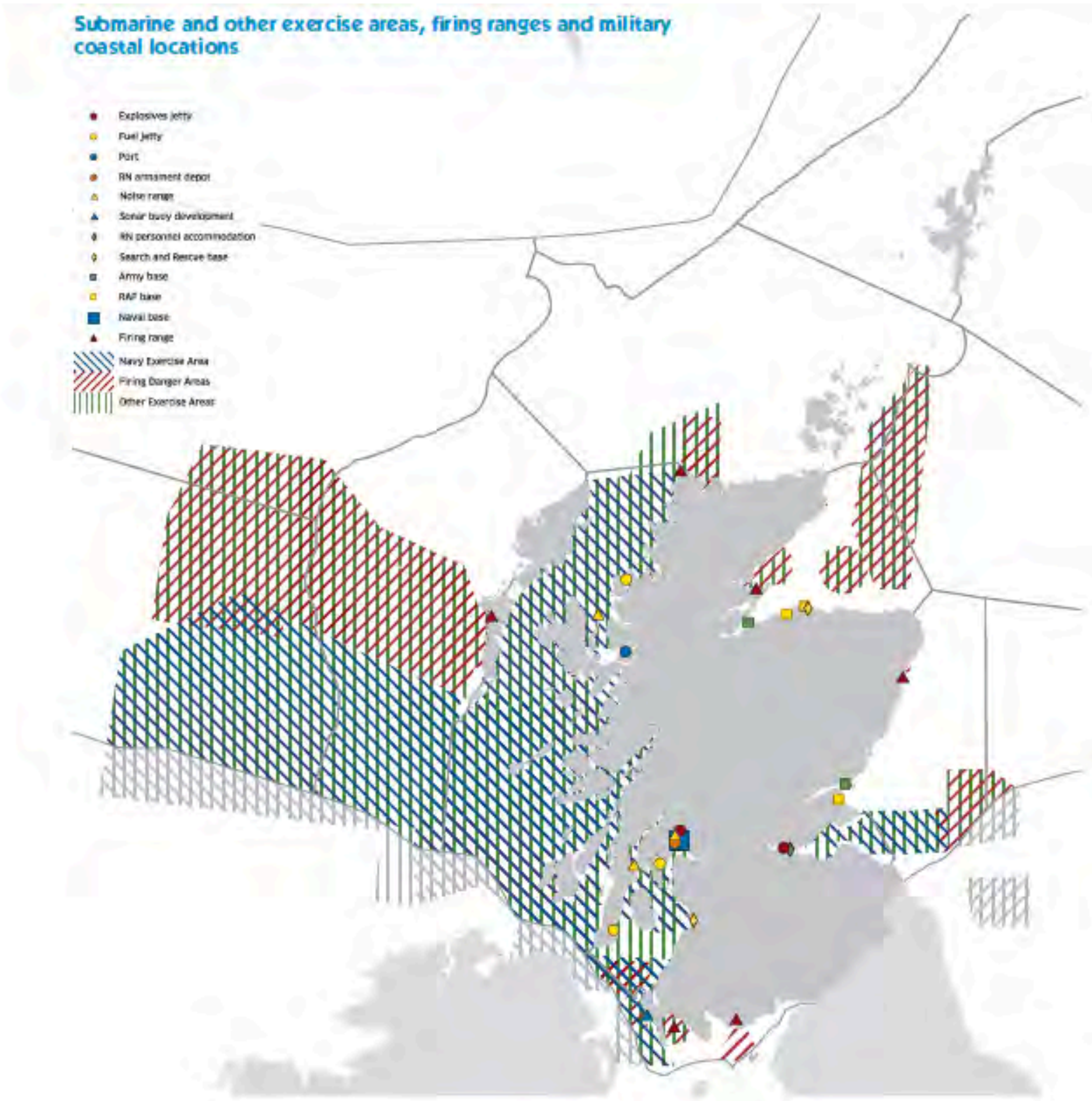


Fig. 44. Submarine and other exercise areas, firing ranges and military coastal locations in Scotland (Source: Marine Scotland)

Although the exercises off North and North-east Scotland have largely been ones that have not involved the use of active sonar for long distance submarine detection, in spring 2015 the Joint Warrior exercise extended from the west of Scotland to the Pentland and Moray Firths, leading to conservation concerns for marine mammals expressed by environmental groups.

Despite all the mid-frequency sonar activity that has taken place in recent decades around the British Isles, there are few cases of potential impacts upon cetaceans. Concerns for possible effects of Naval sonar activities on minke whale and harbour porpoise in West Scotland were reported by Parsons *et al.* (2000), following sharp decreases in sighting rates observed during the exercises.

Between January and July 2008, 18 Cuvier's beaked whales, four Sowerby's beaked whales, five unidentified beaked whales and 29 long-finned pilot whales were reported stranded in Scotland, Ireland and Wales (Dolman *et al*., 2010). Most carcasses were too decomposed for necropsy. Although the initial stranding of five Cuvier's beaked whales in Scotland shared some similarities with atypical mass stranding events linked in time and space to mid-frequency naval sonars, there were two important differences with the remaining strandings during this period. First, the geographical range of the event was very wide, and second, the strandings occurred over a prolonged period of several months. Both of these factors could be related to the fact that the mortalities occurred offshore and the carcasses drifted ashore. The cause(s) of this high number of strandings of mixed offshore cetacean species during this period remain undetermined.

In June 2008, there was a mass stranding of 26 common dolphins in the Fal Estuary, Cornwall, UK (Jepson and Deaville, 2009). All animals examined were in good nutritive status and had empty stomachs. There was no evidence of significant infectious disease or acute physical injury; levels of organochlorines, trace metals and butyltins were relatively low; the ears were grossly normal (but mild decomposition prevented further investigation); there were no signs of gas or fat emboli; and boat strike, by-catch, attack from killer whales or bottlenose dolphins feeding unusually close to shore immediately before the mass stranding, ingestion of harmful chemical or algal toxins, abnormal weather/climatic conditions and high-intensity acoustic inputs from seismic airgun arrays and natural sources (e.g. earthquakes) were all excluded as likely causes. An international naval exercise using mid-frequency active sonar was conducted in the South Coast Exercise Area prior to the mass stranding. However, there were c. 60 hours between the cessation of its use and the stranding and was therefore considered too temporally remote to have directly triggered the event, although they may have played a part in a behavioural response that ultimately led to the stranding. Ultimately, a definitive cause of the mass stranding could not be identified.

Table 11. Characteristics of Active Sonar (from Evans & Miller, 2004)

Sonar Type	Freq. (kHz)	Level (dB/µPa)	Pulse duration	Repetition rate	Beam width (v/h)
a) Short-range Imaging					
Side-scan	36-500	220-230	<0.1ms	variable	35°/2.7°
Multi-beam	15.5	236-238	20ms	15s	4.3°/120°
b) Long-range Detection					
LFA	0.05-0.5	240	6-100s	360-900s	11°/360°
TVDS LF	0.45-0.7	214-228	2+2s	60-90s	23°/360°
TVDS MF	2.8-3.3	223-226	2+2s	60-90s	24°/360°
AN/SQS	2.6-8.2	223-235+	0.5-2s	26s	40°/360°

Since active sonar is used widely in fisheries and oceanographic surveys, it is often questioned why these may not have similar effects to military sonar. However, their beam width is much narrower than those used for long-range detection (Table 11).

3.5.7 Ship Strikes

Although historical records of collisions between ships and cetaceans exist at least as far back as the early 17th century, numbers of cases reported appear to have increased markedly from the 1950s onwards, corresponding to the period when vessels regularly attained speeds of 14-15 knots or more (Laist *et al*., 2001; IWC, 2008). Whales may be hit either by the bow, the keel or any other part of a vessel`s hull, or by its propeller. Hit whales at times may be stuck on the bow of large ships and are often brought into a harbour, sometimes after carrying the carcass over substantial distances (e.g. Laist *et al*., 2001; Pesante *et al*., 2002).

Northwest European seas contain some of the busiest waterways in the world (see Fig. 45). The North Sea receives more than 400,000 ship movements a year, with particularly heavy traffic through the traffic separation scheme in the Strait of Dover where approximately 150 ships per day pass in each direction, in addition to an average 300 ferry crossings daily (North Sea Task Force, 1993). The dredged entrance route to Rotterdam/Europort and its connecting route through the Channel permit navigation of vessels of up to 400,000 tonnes with a maximum depth of 24 m. There is also a heavy flow of shipping from the North Sea to the Baltic via the Kiel Canal, with c. 47,000 vessel movements. Most of the European Community's largest ports are on the North Sea coasts and rivers: Hamburg, Amsterdam, Rotterdam, Antwerp, Le Havre and London. Rotterdam/Europort is by far the largest port, followed by Antwerp, Hamburg, and London. Other areas within the ASCOBANS Region also receive shipping traffic, although the relative densities of these are not clearly known. Over the last twenty years, the numbers of shipping movements, sizes of vessels and their average speeds have all increased in the region (OSPAR, 2010).

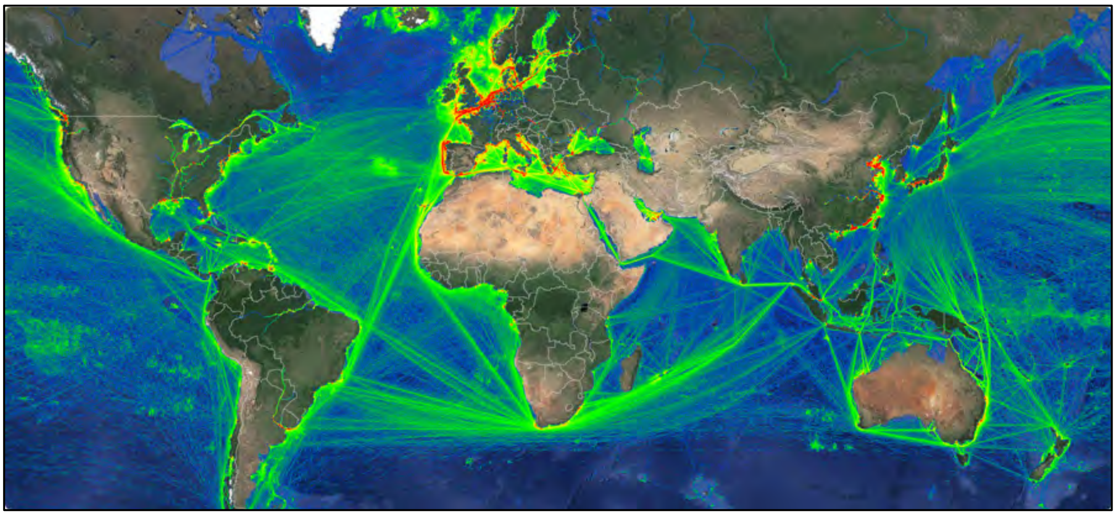


Fig. 45. Satellite derived map of global shipping movements

Approximately half the shipping activity in the North Sea consists of ferries and roll-on/roll-off vessels on fixed routes, while, for example, in United Kingdom ports, tanker traffic represents about 10% and chemicals around 4% of ship departures (North Sea Task Force, 1993).

With the ever greater speeds exhibited by shipping – tankers, ferries, yachts, and a wide variety of small craft, the problem of vessel strikes is likely to increase. In a wide-ranging review of the topic, Laist *et al*. (2001) noted that although all types and sizes of vessels can be involved, most lethal or severe injuries are usually caused by ships travelling 14 knots (26 km/h) or faster and of 80 metres length or more. An analysis of average speeds travelled by vessels in NW European waters, tracked using AIS, indicates that most are travelling at speeds exceeding 10 knots (Figure 47) (Evans *et al*., 2011). Damage in the form of cuts to the dorsal fin and back tend to be the result of strikes from small craft, although larger vessels can also cause similar damage. The probability of a ship strike being lethal increases markedly as vessel speeds increase from 10-15 knots (Fig. 46; Vanderlaan and Taggart, 2007). Evidence of vessel collisions has been reported for at least 21 cetacean species (Evans, 2003).

Since 1990, the UK has been undertaking regular post mortem studies of cetaceans stranding around the British Isles under the Cetacean Strandings Investigation Programme (CSIP). Causes of mortality have been assessed, resulting in estimates of the proportions of post mortem examinations (PMEs) that can be attributed to physical trauma. This excludes animals showing signs of physical damage attributable to either bottlenose dolphin attack or by-catch. However, it includes cases of physical trauma of unknown origin and some of these could belong to one or other of those categories.

Table 12. Cases where physical trauma was diagnosed as the most likely cause of death for cetaceans stranded around the British Isles, 1990-2010 (from Evans *et al*., 2011, analysed from CSIP database)

Cetacean Species	Number of PMEs	Number with physical trauma	Percent with physical trauma
Fin whale	5	1	20%
Minke whale	20	3	15%
Harbour porpoise	1729	76	4%
Common dolphin	346	15	4%
White-beaked dolphin	52	3	6%
Risso’s dolphin	20	1	5%
Sowerby’s beaked whale	16	1	6%

The results indicate that between 15-20% of baleen whales examined at post mortem have suffered mortality from physical trauma whereas in small cetaceans, it is rather less, at between 4-6% (Table 11). Nevertheless, it does highlight that small cetaceans do also experience vessel strike, some having clear signs of blunt trauma including propeller cuts. It is also not confined to just a few species. A review of post mortem results from each country’s strandings programmes reveals that a further seven cetacean species in Northwest Europe have died as a result of physical trauma presumed to be vessel strike: humpback whale, sperm whale, killer whale, long-finned pilot whale, bottlenose dolphin, Atlantic white-sided dolphin, and striped dolphin (Evans *et al*., 2011).

Two methods typically have been used to plot shipping movements:

1) Automatic Identification System (AIS). This is a VHF broadcast system (working on 161.975 MHz and 162.025 MHz) that sends information at regular intervals including the identity of the vessel (MMSI number), its position, course and speed to other vessels and to shore receivers. Since it is a VHF system, transmissions to shore stations (or other vessels) are generally limited to line of sight. Since January 2005, the International Maritime Organization's (IMO) International Convention for the Safety of Life at Sea (SOLAS) required AIS to be fitted aboard international

voyaging ships with gross tonnage (GT) of 300 or more tons, and all passenger ships regardless of size. Within the EU, fishing vessels with an overall length of more than 15 metres were also required to use AIS by 2014. It is estimated that more than 40,000 ships currently carry AIS class A equipment.

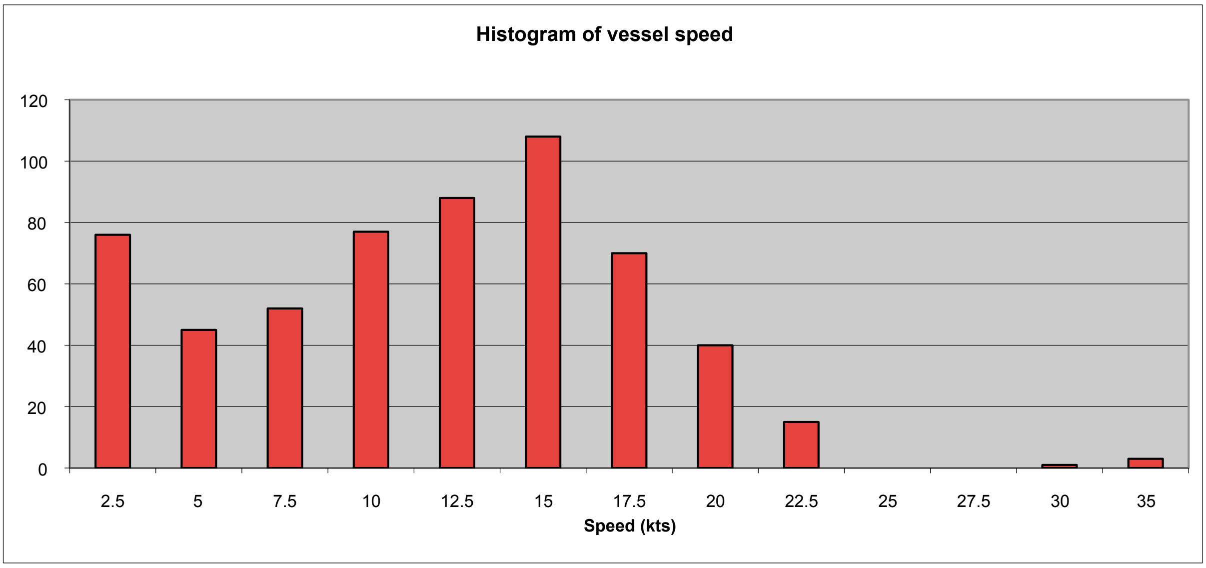
Normally, vessels with an AIS receiver connected to an external antenna placed on 15 metres above sea level, will receive AIS information within a range of 15-20 nautical miles. Base stations at a higher elevation, may extend the range up to 40-60 nm, depending on elevation, antenna type, obstacles around antenna and weather conditions. The most important factor for better reception is the elevation of the base station antenna. The higher it is, the better. Vessels can be detected 200 nm away, with a small portable antenna placed on an island mountain at 700 metres altitude. However, often the receivers are closer to sea level and coverage is much lower, whilst range can be affected by atmospheric conditions. Data can be derived from www.marinetraffic.com//ais, which has c. 200 AIS receivers within Northwest Europe.

2) VOS Monitoring Systems. Ships from many countries voluntarily participate in collecting meteorological data globally, and therefore also report the location of the ship. Such data can be used to map shipping densities), and have been utilized to identify areas where shipping noise may be a particular threat to marine mammals (NMFS, 2005; AEI, 2010). Evans *et al*. (2011) used data collected from October 2004 – September 2005 (as part of the World Meteorological Organization Voluntary Observing Ships Scheme; http://www.vos.noaa.gov/vos_scheme.shtml; see also Halpern *et al*., 2008). This year was chosen as it had the most ships with vetted protocols and so should provide the most representative estimate of global ship locations.

Because the VOS program is voluntary, much commercial shipping traffic is not captured by these data. Therefore, estimates of the shipping are biased (in an unknown way) to locations and types of ships engaged in the programme. In particular, high traffic locations may be strongly underestimated, although the relative impact on these areas versus low-traffic areas appeared to be well-captured by the available data (Fig. 47; see Evans *et al*, 2011), and areas identified as without shipping may actually have low levels of ship traffic. Furthermore, because ships report their location with varying distance between signals, ship tracks are estimates of the actual shipping route taken.

Bearing in mind the limitations of data collection on shipping that each method has, nevertheless, both AIS and VOS data agree with one another, highlighting the following areas as having high shipping densities: English Channel, southernmost North Sea, Kattegat and Danish Belt Seas, and western and central Baltic. Large cetaceans, the group identified as most vulnerable to ship strike, are comparatively scarce in all those areas, with all the species except minke whale occurring mainly in deep waters off the edge of the continental shelf. For the large whales, the shelf edge, Bay of Biscay, NW Spain and north-western North Sea have the highest densities, at least between April and December; when shipping densities are incorporated into the models, the main areas of strike risk are parts of the Celtic Sea, Bay of Biscay, and off NW Spain (Fig. 48; see also Evans *et al*., 2011). As for the minke whale, which in summer is largely a shelf species, most abundant at that time in the north and west of the British Isles, spatial overlap with shipping is likely to be relatively small. For that species, the only areas with some strike risk would be the central west North Sea and western English Channel, and some localised parts of the Irish Sea.

a)



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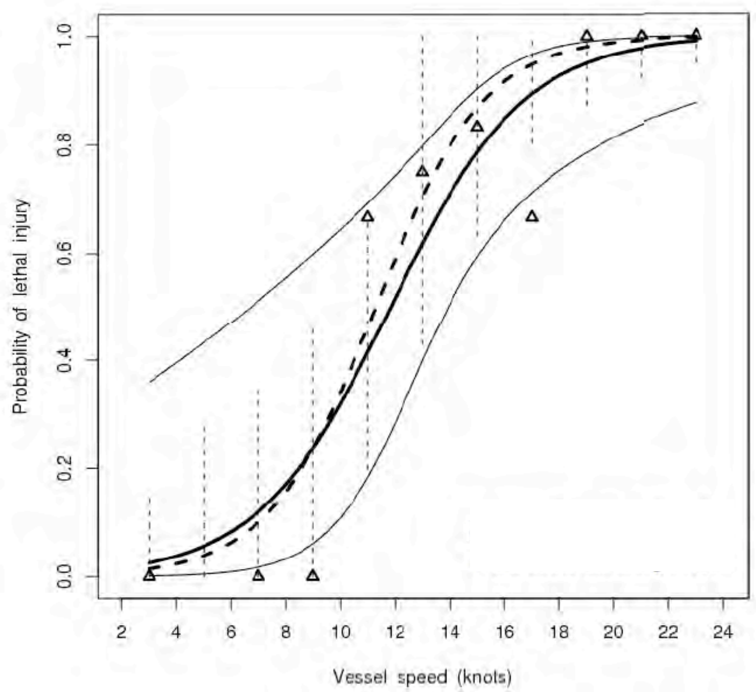

Fig. 46a) Frequency of Vessel Speeds amongst Shipping in NW Europe (source: Evans *et al*, 2011); and b) Probability of a Lethal Strike at different Vessel Speeds (source Vanderlaan and Taggart, 2007)

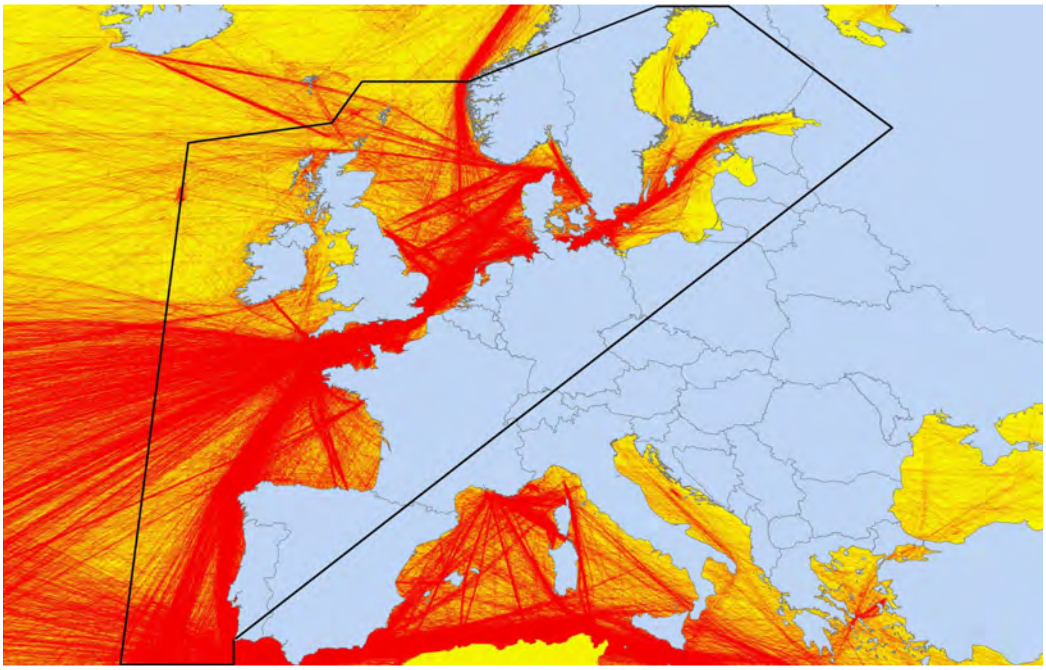

Fig. 47. VOS annual tracks of commercial vessels in Northwest Europe (represented by the area within the black lines) (source: Evans *et al*., 2011)

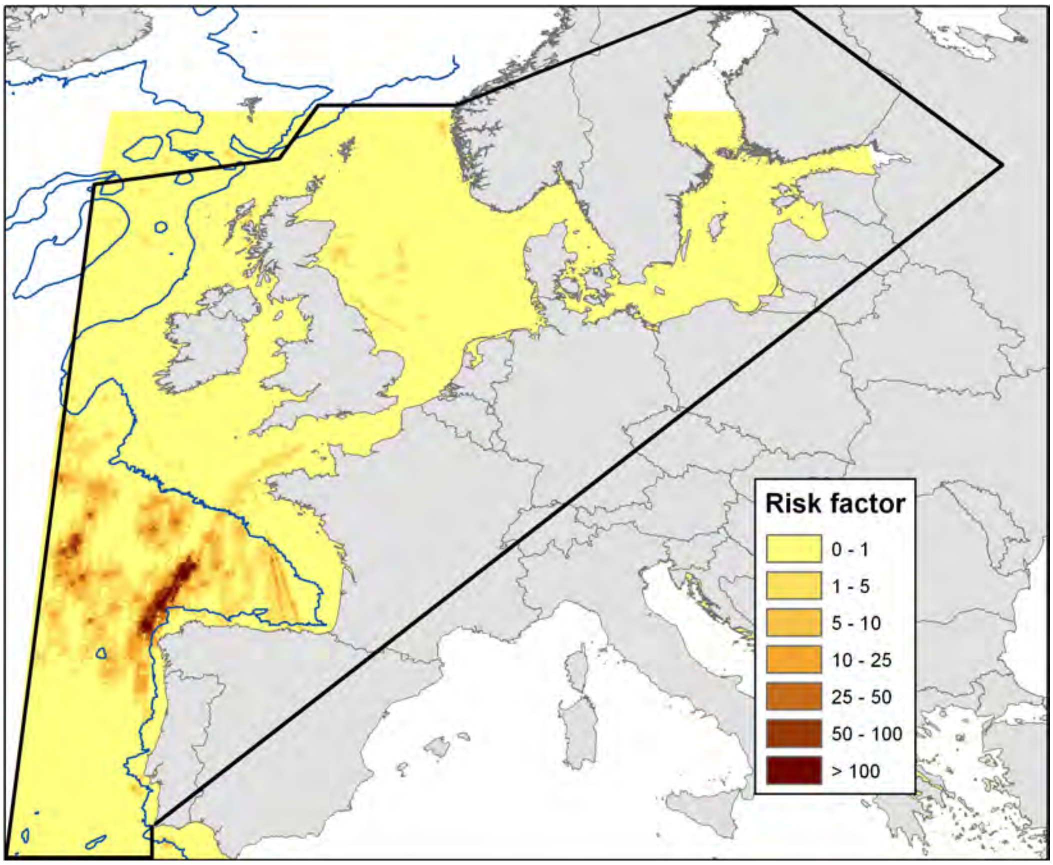

Fig. 48. Potential Risk Areas for the more vulnerable cetaceans in NW Europe (source: Evans *et al*., 2011)

3.5.8 Recreational Disturbance

Coastal areas are popular for leisure and recreation, attracting both local people and tourists from inland and abroad. Activities include bathing, surfing, sailing, sea angling, water sports, and wildlife watching (marine mammals, seabirds, and basking sharks). Tourism in North-west Europe is distinctly seasonal, with overnight stays concentrated in the summer months (OSPAR, 2010). In all parts of the region, tourism has been growing steadily; in the Republic of Ireland, for example, it has been estimated that since the 1970s, the number of day trips to the coasts has increased by almost 600% (OSPAR, 2010).

As recreational activities increase in coastal zones around the world, pressures upon a number of cetacean species have increased, leading to concerns expressed in many areas (Evans, 1996; Würsig and Evans, 2001; Williams *et al*., 2002, 2006; Bejder and Samuels, 2003; Constantine *et al*., 2004; Lusseau, 2004; Lusseau and Higham, 2004; Lusseau *et al*., 2006; New *et al*., 2013; Higham and Bejder, 2014).

Table 13. Estimates of numbers and income from whale watchers in Europe, 1991-2008. AAGR = Average annual growth rate (Source: Connor *et al*., 2009)

Year	Number of whale watchers	AAGR	Number of countries	Direct expenditure	Indirect expenditure	Total expenditure
1991	158,763	N/A	8	$2,161,000	$3,429,000	$5,690,000
1994	204,627	8.8%	16	$4,123,000	$17,862,000	$21,985,000
1998	418,332	19.6%	18	$11,048,000	$34,981,000	$46,029,000
2008	828,115	7.1%	22	$32,346,906	$65,290,135	$97,637,041

Whale watching in particular has increased dramatically in many parts of the world. The latest comprehensive estimate for Europe by Connor *et al*. (2009) found that numbers had doubled in ten years since Hoyt's (2001) estimate (Table 12), averaging 7% growth per annum. This increase was somewhat surprising for a region with a mature tourism industry. Over that decade, whale watching in Europe had expanded by four new countries to a total of 22 countries, generating annually nearly $100 million in expenditure, from Cyprus to Greenland. Europe accounted for 6% of global whale watchers (O'Connor *et al*., 2009). This increase seems to have continued apace since the last global estimate. In Wales, for example, 35,000 visitors going out to see bottlenose dolphins on trip boats generated an estimated £2.6 million income a year overall through ticket sales, local accommodation, purchase of food and merchandise (O'Connor *et al*., 2009), and since then, the estimated annual numbers has increased further (by 35 percent) to almost 50,000, with an associated 65 percent increase in direct revenue from ticket sales from 863,000 in 2008 to 1.43 million US dollars in 2011 (O'Connor *et al*., 2009; Lambert and Evans, 2012).

Other forms of active marine-based recreation include sea kayaking, sailing, sea angling and water sports such as power boating, personal water craft (jet skis), water skiing, and wind surfing (OSPAR, 2008).

Cetacean species most likely to be affected by recreational activities are those frequenting the coastal zone. In North-west Europe, these are harbour porpoise and bottlenose dolphin, and in some areas, minke whale, white-beaked dolphin, short-beaked common dolphin, Risso's dolphin and killer whale. Needless to say, these also are the species most often targeted by whale and dolphin watchers, although if a rarer species like humpback or fin whale turns up, commercial whale watching may start opportunistically, stopping once the animal(s) moves on.

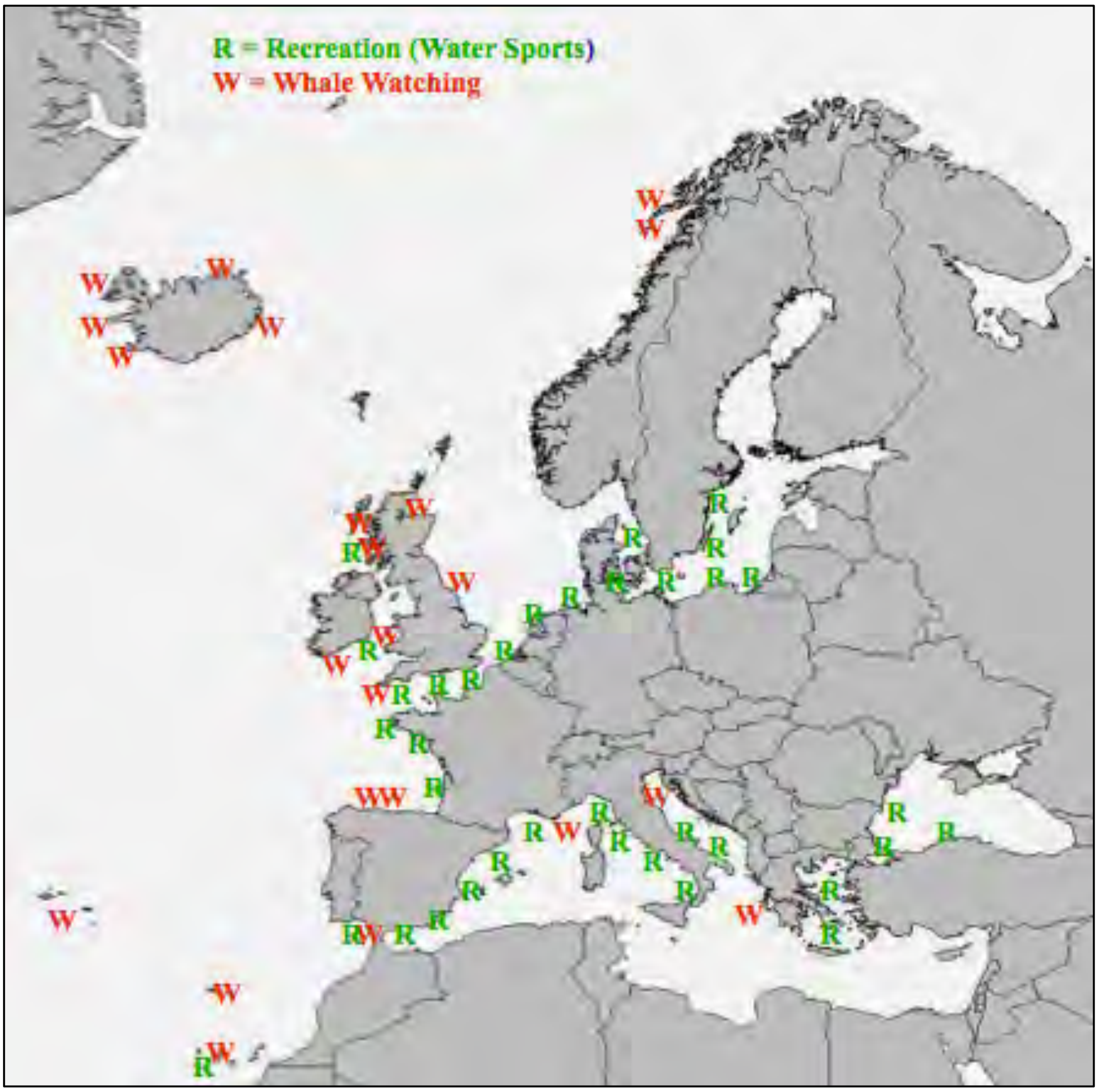

Fig. 49. Map of distribution of main areas where marine recreation and whale and dolphin watching take place

The main areas where marine recreational activities and whale watching occur are shown in Figure 49. Whale watching in Northwest Europe is concentrated around the British Isles, into five regions: the Moray Firth in NE Scotland where bottlenose dolphin is the target species; the Hebrides of West Scotland where a variety of species may be seen but the main species targeted are minke whale, common dolphin, and Risso's dolphin; West Wales with bottlenose dolphin targeted in Cardigan Bay and harbour porpoise and common dolphin around the Pembrokeshire islands; West Cornwall where bottlenose dolphin, common dolphin and Risso's dolphin tend to be the target species; and eastern England off the coasts of North Yorkshire and Northumberland where white-beaked dolphin, minke whale and larger whales like fin and humpback are targeted opportunistically. In the Republic of Ireland, there is whale watching off the south-west coast, aimed primarily at humpback whales, although other species regularly seen include minke whale, common dolphin and harbour porpoise. There is no regular whale watching in the southern North Sea, and rather little around the coasts of France, confined to some bottlenose dolphin watching on an opportunistic basis off the coasts of Normandy and Brittany.

Other recreational activities (sailing, kayaking, water sports) are much more widespread, occurring all along the coasts of northern Germany, the Netherlands and Belgium as well as the south coast of England, and at scattered localities in Southwest England, Wales, Ireland and Southwest Scotland. Marine recreation in eastern Britain takes place mainly in the southeast.

The presence of vessels can have both direct and indirect effects on cetaceans (Nowacek *et al.,* 2001; Mattson *et al.,* 2005; Lusseau, 2006). They can cause disturbance of feeding activities, separate calves from their mothers, and interfere with acoustic communication, whilst physical contact may lead to injury or death. Short-term effects include changing behavioural patterns such as increased swim speeds (Au and Perryman, 1981; Kruse, 1991; Nowacek *et al.,* 2001; Williams *et al.,* 2002*;* Mattson *et al.,* 2005), increased dive intervals (Janik and Thompson, 1996; Nowacek *et al*., 2001; Williams *et al*., 2002, 2006; Constantine, 2004; Lusseau, 2003b, 2006), greater breathing synchrony (Hastie *et al*., 2003), vertical and/or horizontal evasion (Nowacek *et al*., 2001; Williams *et al*., 2002, 2006; Hastie *et al*., 2003; Mattson *et al.,* 2005; Feingold and Evans, 2014), reduced inter-animal spacing (Bejder *et al.,* 1999; Nowacek *et al*., 2001), and changes in behavioural state such as reduced resting behaviour (Lusseau, 2003a; Constantine *et al*, 2004). In the presence of vessels, cetacean echolocation and vocalisations have the potential to be masked or altered (Hastie *et al*., 2003; Buckstaff, 2004; Mattson *et al*., 2005; Thompson, 2012), and this may affect group cohesion (Nowacek *et al*., 2001; Constantine *et al*., 2002; Mattson *et al*., 2005; Thompson, 2012; Richardson, 2012).

Longer-term effects that may have population consequences can be changes in residency patterns (Lusseau, 2005; Bejder *et al*., 2006a, b; Feingold and Evans, 2014), a reduction in population size due to the suppression of reproductive capabilities and/or a reduction in the consumption of prey leading to reduced energy intake (Williams *et al.,* 2004, 2006; New *et al*., 2013; Pirotta *et al*., 2015). Additionally, migration, reduction in usage and/or long-term abandonment of favoured sites may occur in highly disturbed areas (Kruse, 1991; Nowacek *et al*., 2001; Lusseau, 2004; Bejder *et al*., 2006a, b; Pierpoint *et al*., 2009; Feingold and Evans, 2014).

In the region under investigation here, there is evidence of negative effects upon bottlenose dolphins in Cardigan Bay, West Wales (Pierpoint *et al*., 2009; Richardson, 2012; Thompson, 2012; Feingold and Evans, 2014) and in the Moray Firth, Northeast Scotland (Janik and Thompson, 1996; Hastie *et al*., 2003; New *et al*., 2013; Pirotta *et al*., 2015).

3.5.9 Climate Change

Whereas the Earth's climate has exhibited broad extremes over geological time there is general consensus that the rate of global warming is unprecedented in both terrestrial and marine environments (IPCC, 2007). Ocean climate is largely defined by its temperature, salinity and ocean circulation, and the exchange of heat, water and gases (including CO_2) with the atmosphere. The functioning of marine ecosystems is highly dependent on changes to both ocean climate and acidification, whilst storms and waves, sea level rise and coastal erosion pose clear threats to human life as well as to other creatures (MCCIP, 2010).

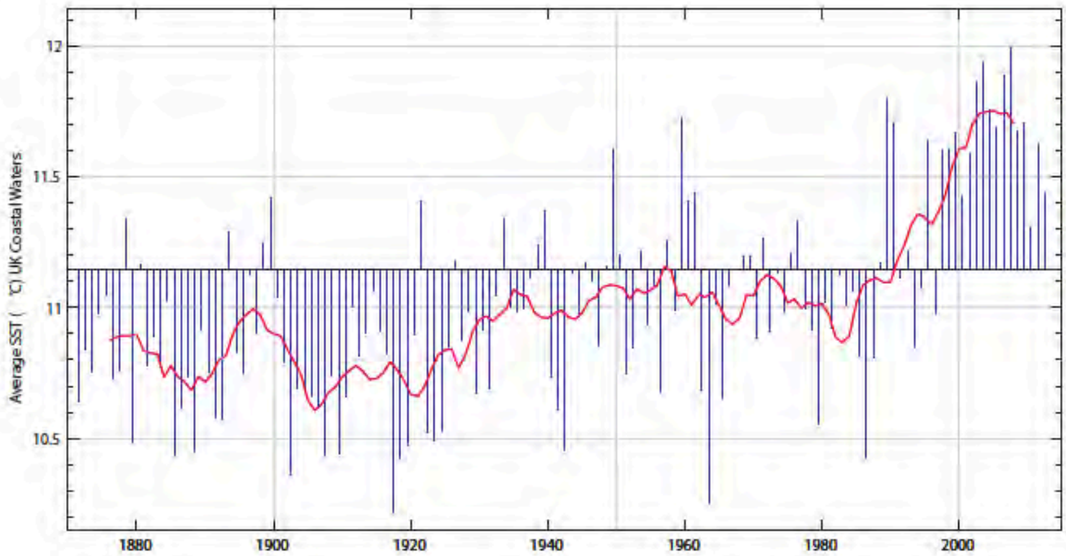

Fig. 50. Time series of average SST in UK coastal waters, 1872-2012 (source: Dye *et al.*, 2013). The blue bars show the annual values relative to the 1971-2000 average and the smoothed red line shows the 10-year running mean. Data are from the HadISST1.1 data set (Rayner et al., 2003)

Marine air and sea temperatures have risen over the northeast Atlantic in the last 25 years (Fig. 50), the largest increase in sea surface temperatures occurring in the southern North Sea and eastern English Channel, at a rate of between 0.6 and 0.8^{o} C per decade (Rayner *et al.*, 2003; Dye *et al.*, 2013; see Fig. 51). Although inter-annual variability is high, the first decade of the 2000s was the warmest on instrumental record (IPCC, 2007; Hughes *et al.*, 2010). The rate of change in ocean pH is thought to be faster than anything experienced in the last 55 million years, with a 30% decrease in pH, and a 16% decrease in carbonate ion concentrations since 1750 (Caldeira and Wickett, 2003; IPCC, 2007; Doney *et al.*, 2009).

Although the Arctic is characterized by large temporal and spatial variations in climate, the past few decades have seen record minima in sea ice coverage during summer and increased melt from Greenland, which has exceeded the range of natural variability over the past thousand years (Morison *et al.*, 2000; ACIA, 2004; IPCC, 2007; Walsh, 2008).

Relative to the underlying warming trend during the 20th century, the surface waters averaged over the north Atlantic were cool in the period between 1900 and 1930, warm from 1930 to 1960, cool between the late 1960s and 1990 and then warm from 1990 to present (Dye *et al.*, 2013). Warming due to anthropogenic effects is superimposed onto this pattern of multi-decadal variability, which is thought to be a natural pattern variation and has been described as the North Atlantic Oscillation

(NAO) or Atlantic Multi-decadal Oscillation (AMO) (Knight *et al*., 2005). Whilst it is clear that there is a significant multidecadal pattern to sea-surface temperatures, there is still much uncertainty about how to determine the relative contribution of these two factors to the recent observed warming (Knight *et al*., 2005; Cannaby and Hüsrevoğlu, 2009; Swanson *et al*., 2009; Ting *et al*., 2009).

Unstable weather patterns leading to increased frequency of cyclones and other types of storm also appear to be influenced at least in part by the North Atlantic Oscillation (the index of which is a measure of the difference in mean atmospheric pressure between high pressure in the Azores (or Gibraltar) and low pressure in Iceland). The recent strong trend in the NAO (towards stormier conditions) is apparently unique in its history, but it is still under debate as to whether or not this is a response to greenhouse gas forcing (Osborn, 2004).

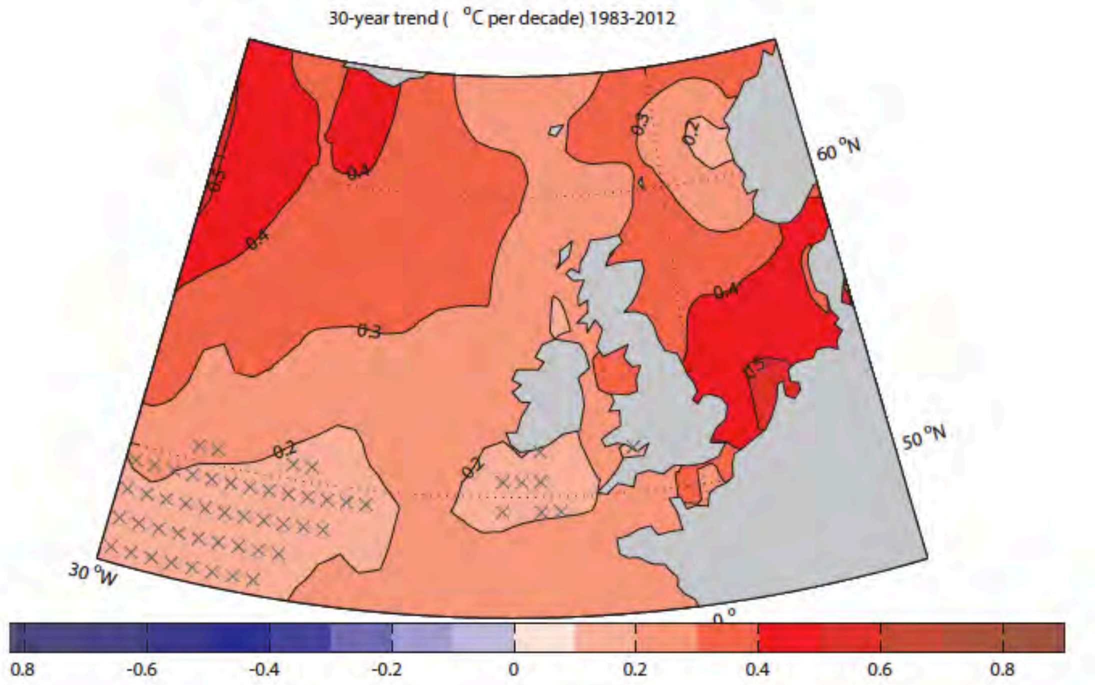

Fig. 51. Trend in annual average sea-surface temperature (°C/decade) from 1983 to 2012 (Source: Dye *et al*., 2013). Data are from the HadISST1.1 data set (Rayner *et al*., 2003). Hatched areas have a slope, which is not significant at the 95% confidence level (alpha=0.05) using Mann-Kendall non-parametric test for a trend

More general changes in westerly winds in the North Atlantic region are implicated in changes in wave heights and storminess around Western Europe, and the behaviour of the NAO is not the only relevant factor. Another pattern of atmospheric pressure anomalies, the East Atlantic Pattern (EAP), appears to explain a large part of the inter-annual variability in winter wave climate in the region, where significant increases have taken place between the 1960s and early 1990s (Woolf *et al*., 2002).

Increased sea surface temperatures have led to extensive changes in plankton communities. In the North Sea, the population of the previously dominant and important zooplankton species, the coldwater copepod *Calanus finmarchicus*, has declined in biomass by 70% since the 1960s. Species with warmer-water affinities (e.g. *Calanus helgolandicus*) are moving northward to replace it, but are present at much lower abundance (Fig. 52). This could have far-reaching consequences upon the higher trophic levels.

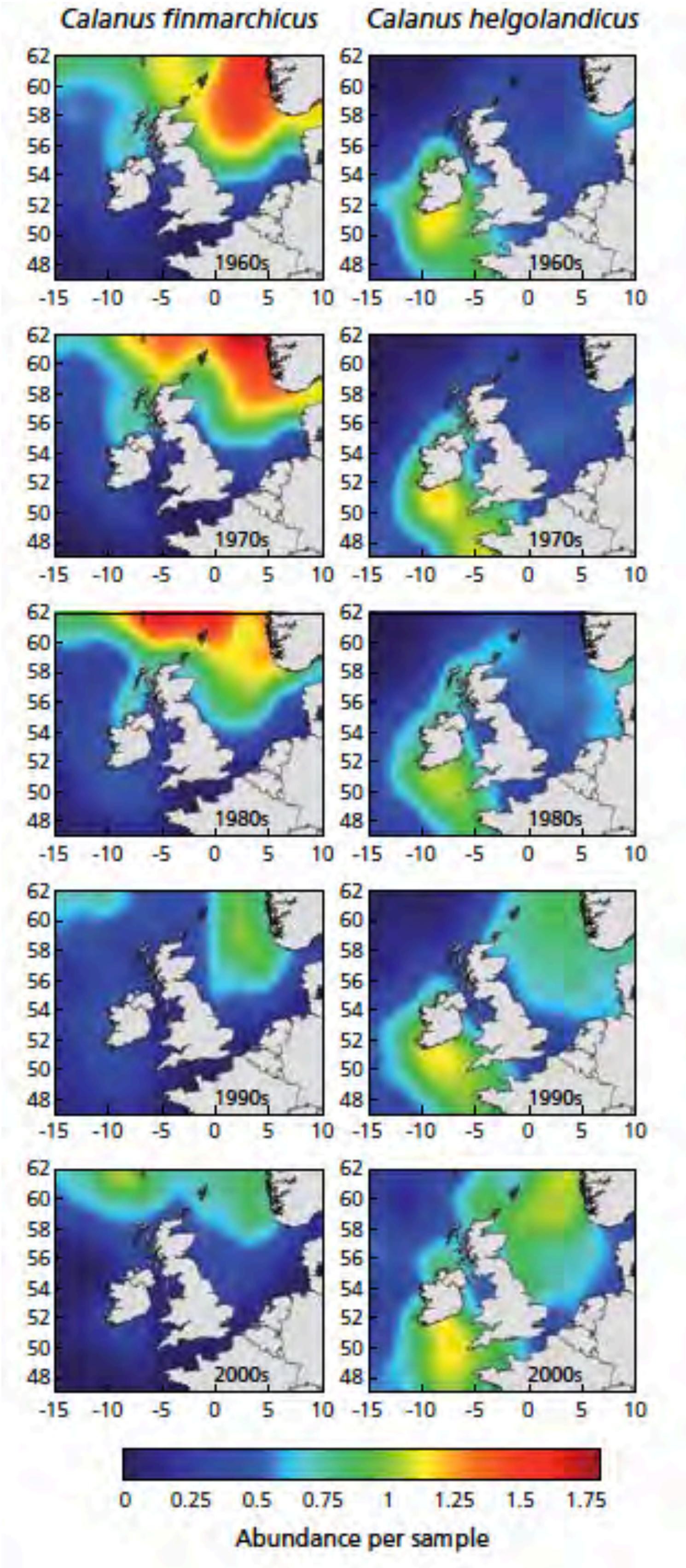

Fig. 52. Changes in the mean decadal abundance of *Calanus finmarchicus* and *Calanus helgolandicus* in the North-East Atlantic (Source: Defra, 2010, from Continuous Plankton Recorder Survey)

As an example, the life cycle of the sandeel is timed to make use of the seasonal production of copepods, which in turn depend upon planktonic plant production. Not only has copepod abundance declined but the spring occurrence of copepods and fish larvae has become out of synchrony resulting in low recruitment of young sandeels. When this occurs, it can affect top predators such as seabirds (Daunt and Mitchell, 2013) and marine mammals (Evans and Bjørge, 2013).

A number of reviews have been published recently of the possible effects of climate change upon marine mammals (IWC, 1997, 2009; Würsig *et al*., 2001; Learmonth *et al*., 2006; Huntington and Moore, 2008; Laidre *et al*., 2008; MacLeod, 2009; Evans *et al*., 2010a, b; Evans and Bjørge, 2013). Marine mammals, as warm-blooded thermo-regulating vertebrates, might be expected to cope well with most environmental variation predicted from climate change. They employ complex behavioural adaptations that can lead to them having strong buffering against environmental variability, including variation in food supply. These adaptations can extend to life-history processes, some of which are sensitive to temperature, especially with respect to the thermoregulation of neonates. On the other hand, changes in the availability of their habitat (including food resources) may lead to changes in population size or distribution in particular cases. The most obvious example in this context is the reduction in ice cover affecting ice-breeding polar seals such as the walrus, bearded, hooded, ribbon, harp or ringed seal, and its consequent effect upon Arctic predators like the polar bear (Stirling *et al*., 1999; Derocher *et al*., 2004; Ferguson *et al*., 2005; Huntington and Moore, 2008).

Arctic cetaceans like narwhal, beluga and bowhead whale are likely to be most affected by climate change as these associate closely with ice (Huntington and Moore, 2008; Laidre *et al*., 2008; IWC, 2009). In North-west Europe, one might expect an increase in the number of species occurring, with range extensions from subtropical and tropical regions as sea temperatures rise, and that is indeed what is being experienced – striped dolphin and Cuvier's beaked whale are being recorded more regularly and dwarf sperm whale has been added to the British fauna (Evans *et al*., 2010a; Evans and Bjørge, 2013). On the other hand, cold-water shelf species like the harbour porpoise and white-beaked dolphin, and shelf edge species like the Atlantic white-sided dolphin may find conditions less favourable, and their ranges could start to contract at the southern margins (MacLeod, 2009; Evans *et al*., 2010; Evans and Bjørge, 2013).

As noted above, most effects upon cetaceans are likely to be seen through their food supply. In the North Sea, climate change impacts are predicted in fish species like sand eel and sprat, recruitment in the former being negatively affected whereas the latter may benefit from warming sea temperatures (Pinnegar and Heath, 2010). This might explain the southwards shift in harbour porpoises in the western North Sea since the 1990s if sand eel recruitment has been poor in recent years (Pinnegar and Heath, 2010; Hammond *et al*., 2013).

4. DISCUSSION

Relatively good regular spatial coverage in the study area has only taken place in the last ten years, and even then it is by no means comprehensive. This has meant that establishing trends particularly on a regional basis and examining robustly for relationships with seismic activities remains a major challenge. Given also that there are several other human activities that may have impacts on cetacean populations, disentangling these to establish whether or not oil and gas exploration has played a role in any observed status changes is even more challenging.

Statistical data for many of the human activities that can impact upon cetaceans over the time period being examined are not available. For that reason, it has not been possible to incorporate these into a generalised additive model to assess their relative importance. We are therefore forced to assess this qualitatively. We have attempted to do this in Table 13, for the ten main species occurring in NW Europe.

Drawing upon our present knowledge from the literature as described in section 3.5, for most species there is little evidence that anthropogenic activities are likely to have a major negative impact in the region, although it is possible in some cases that recent distributions have been shaped by earlier negative interactions with some.

Each of the ten main species will be reviewed in turn:

a) *Fin whale* This species experienced sustained hunting pressure until the 1970s since when there is some indication that the North Atlantic population is recovering. Its predominantly deepwater distribution beyond the continental shelf means that it scarcely encounters a number of the human activities occurring predominantly in the more coastal areas of the region, e.g. recreation, marine construction noise. Ship strike is probably the major negative pressure. If its favoured prey were to increase within the region, then one might expect the species also to increase. Most seismic survey effort has been in the North Sea where the species is uncommon; seismic exploration along the shelf edge generally did not increase until the mid 1990s. If there is a negative impact from this activity, it is unlikely to be detected during the period of this study, but may be in the future.

b) *Minke whale* The commonest baleen whale on the NW European continental shelf, several lines of evidence have indicated that the minke whale has increased over the last thirty years. Of all the whale species occurring in the region, this is the one that might be predicted to be negatively affected by seismic exploration given the co-occurrence of the two in the North Sea. However, there is no evidence of a negative relationship between its distribution in the different decades and that of seismic activity. Although hunting continues in neighbouring Norwegian seas, catches are much lower than they were before the 1980s. Other potential pressures include bycatch, shipping noise, local disturbance from marine recreation, and prey depletion both directly from overexploitation by fisheries of target species and indirectly if climate change has negative effects on favoured prey. So far, however, none of these appear to be sufficient to reverse an upward trend that may be the result of reduced direct mortality from earlier decades of hunting.

c) *Killer whale* There is no evidence to suggest that this specoes was ever common in UK waters, occurring mainly offshore and with the largest populations known to be north of the British Isles. The recovery of mackerel and herring stocks following over-exploitation in the 1960s and the increase in seal (mainly grey seal) populations in some northern regions may be influencing increased sighting rates in the Northern Isles of Britain, as killer whales exploit prey in that region, but there is little to suggest that the species has been affected by potential negative factors such as shipping or seismic noise. Low reproductive rates are typical of this species, and have been linked to high pollutant loads although a cause-effect relationship has yet to be established.

d) *Long-fnned pilot whale* The long-finned pilot whale is another predominantly pelagic species, occurring largely beyond the NW European continental shelf edge. It continues to be hunted in the Faroes, although in relation to the overall size of the North Atlantic population, the current levels of mortality due to this may be insufficient to see a population effect. Other potential negative impacts such as shipping, seismic or sonar noise, and changes in prey abundance appear not to be sufficient to affect distribution trends, although its pelagic nature means that survey coverage has been rather limited in both time and space.

e) *Risso's dolphin* This species is also largely pelagic but more regularly enters shelf seas, for example off west Scotland, in the western English Channel, and in the Irish Sea. Potential negative impacts include shipping, seismic and sonar noise. The last is included because of its deep diving behaviour and the fact that gas/fat emboli (a feature of behavioural responses to active sonar) have been found in the species. Plastic ingestion may also have a negative impact since the species is known to use suction feeding when capturing its cephalopod prey. Changes in cephalopod distributions related to climate change have recently been observed in the North Sea, but in this instance they have actually been positive, with recent incursions from the Atlantic (van der Kooij *et al*., 2016). This probably explains the observed recent range extension of Risso's dolphin into the northern North Sea. Marine recreation may have a local effect in some areas (e.g. northern Hebrides). No distributon changes have been observed over the three decades with nothing to suggest that seismic activities have impacted Risso's dolphins, although its predominantly offshore Atlantic distribution makes it difficult to test with the data available.

f) *Atlantic white-sided dolphin* The Atlantic white-sided dolphin is another pelagic species that occurs primarily beyond the continental shelf edge. It has a predominantly northern distribution and by the latest time period, it appears to be recorded only in the northernmost part around the Northern Isles of Scotland. This could be related to climate change affecting favoured prey species, although that has yet to be demonstrated. The species is known to experience some bycatch, and may have been affected by the overexploitation of some of its fish prey (e.g. herring, mackerel) in earlier decades. Again, there is no evdence to suggest that seismic activities have impacted on the species although the same proviso applies that its offshore Atlantic distribution makes this difficult to test with the data available being concentrated within shelf seas.

g) *White-beaked dolphin* In the study region, this is a species largely of the continental shelf. With a distribution centred upon the North Sea, the white-beaked dolphin may be predicted to be negatively affected by seismic exploration given the co-occurrence of the two here for several decades. However, as with minke whale, there is no evidence of a negative relationship between its distribution in the different decades and that of seismic activity. Other human activities that could potentially have a negative impact include over-exploitation of prey such as herring and mackerel in the first decade or two, shipping noise, and then in the last two decades, possibly climate change if they have had an effect on potential prey. However, the evdence suggests that, although there may have been a range shift northwards out of the southernmost North Sea since the mid-1990s, overall the population appears to have increased. White-beaked dolphins are known to take a variety of fish species such as herring and mackerel that have also increased in recent years.

Table 13 (a-c). Human activities that may negatively impact on cetacean species in NW Europe

a) Fin whale

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Ship strike			
Climate change			

b) Minke whale

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Climate change			

c) Killer whale

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Climate change			

Table 13 (d-f). Human activities that may negatively impact on cetacean species in NW Europe

d) Long-finned pilot whale

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Ship strike			
Climate change			

e) Risso's dolphin

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Climate change			

f) Atlantic white-sided dolphin

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Climate change			

Table 13 (g-i). Human activities that may negatively impact on cetacean species in NW Europe

g) White-beaked dolphin

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Ship strike			
Climate change			

h) Short-beaked common dolphin

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Climate change			

i) Bottlenose dolphin

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Climate change			

Table 12 (j). Human activities that may negatively impact on cetacean species in NW Europe

j) Harbour porpoise

Time Period	**1980s**	**1990s**	**2000s**
Pressure			
Hunting			
Prey depletion			
Bycatch			
Pollution			
Plastic ingestion			
Shipping noise			
Seismic noise			
Sonar noise			
Marine construction noise			
Recreation disturbance			
Ship strike			
Climate change			

Notes

Green = negative impact unlikely; Amber = negative impact possible;
Red = negative impact more likely

h) *Short-beaked common dolphin* The short-beaked common dolphin occurs both along the edge of the continental shelf and in shelf seas though mainly bordering the Atlantic. It is a species of generally warmer waters than the Atlantic white-sided or white-beaked dolphin. The most obvious human activity known to negatively impact on the species is bycatch. Mortality from this source has been relatively high throughout the last two decades. It may have been affected by the overexploitation of some of its fish prey (e.g. herring, mackerel) in earlier decades. Other human activities with potential negative impacts include shipping and seismic noise, and in recent decades, possibly climate chang although some of the prey species that common dolphins are known to target, such as sardine and anchovy, hav actually increased in the study region, and may explain its recent incursion into the northern North Sea. There is no indication that distribution changes have been affected by seismic activities.

i) *Bottlenose dolphin* Bottlenose dolphins occur in NW European shelf seas largely either along the continental shelf edge (where the greatest numbers can be found) or in inshore waters where it may be locally common and resident or semi-resident. As a consequnce, different populations of the species may experience different human pressures. In northern Europe, there is litte evidence of any substantial mortality from bycatch. The coastal populations are exposed to a variety of human activities, the most important of which appear to be pollution, noise from marine construction (e.g. in the development of windfarms), and marine recreation (including dolphin watching), the latter two activities of which have increased markedly in the last decade or two. Other human activities with potential negative impacts include shipping and seismic noise but at least in the case of the latter, there is no significant relationship between any bottlenose dolphin distribution trends and seismic activities over the three decades, although there is some indication that the species is increasing (and expanding its North Sea range) in the last one or two decades.

j) *Harbour porpoise* The harbour porpoise is the most common and widely distributed cetacean species of NW Europan shelf seas. It is the third of the ten species whose distribution coincides well with where most seismic activity has occurred over the last three decades, and yet in the models there is no evidence for any significant relationship between the two. The species in fact appears to have increased since the 1980s. The main known negative impact upon harbour porpoises is mortality from bycatch, although this may have declined somewhat since the 1990s. Other human activities that potentially may have a negative impact include shipping noise, marine construction (particularly of wind turbines snce the 1990s), and marine recreation that has increased over the last two decades. Changes in prey abundance either from fisheries overexplotation or as a indirect result of climate change (which appears to be having a negative effect on some prey species, notably sandeel) may also affect harbour porpoise, and could account for some of the porpoise distributional shifts that have been observed (from the northwestern North Sea southwards).

5. CONCLUSIONS & FUTURE RECOMMENDATIONS

The fact that this study has found no evidence for a negative impact upon cetacean distributions of seismic exploratory activities in NW European seas may be due to any of the following: 1) there has been no long-term impact; 2) the cetacean survey data available are inadequate to demonstrate an impact; and 3) there may have been an impact but it is masked by other strong effects (natural or from other human activities). We are not in a position to evaluate which of these apply, and it may be a combination of them.

The North Sea is the main part of the study region that has experienced decades of seismic activity in exploration for oil and gas resources. Whereas those seismic activities started in the North Sea in the 1960s (reaching a peak in the 1970s), dedicated cetacean surveys in that region were few and far between until the 1990s. Thus if there had been an initial impact, it would not be detected. With seismic activities in the North Sea now declining, additional cetacean survey effort here during the 2010s and 2020s may not provide any more robust test for whether there are long-term effects. However, what can be concluded is that if there was an initial negative impact from a large amount of seismic survey effort, it does not appear to have persisted over several decades. Species like the minke whale, bottlenose dolphin and harbour porpoise could possibly be recovering from earlier effects but there are equally plausible reasons for this being caused by other human activities (hunting in the case of the minke whale, pollution in the case of the bottlenose dolphin, and both bycatch and pollution in the case of the harbour porpoise).

Since cetacean survey effort continues to improve and has been relatively good during the 2010s, whilst seismic activities continue in some regions, a future recommendation is to repeat these analyses for the current decade. It may then also be possible to incorporate spatio-temporal patterns in other human activities in a more quantitative manner, which would permit these to be included in the modelling process.

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APPENDIX 1: List of 35 European Cetacean Species and their Latin Names

ORDER CETACEA

SUB-ORDER MYSTICETI, the Baleen Whales

Family Balaenidae (right whales)	
Balaena mysticetus	Bowhead whale
Eubalaena glacialis	North Atlantic right whale
Family Balaenopteridae (rorquals)	
Balaenoptera acutorostrata	**Minke whale**
B. borealis	**Sei whale**
B. edeni	Bryde's whale
B. musculus	Blue whale
B. physalus	**Fin whale**
Megaptera novaeangliae	**Humpback whale**

SUB-ORDER ODONTOCETI, the Toothed Whales

Family Physeteridae	
Physeter macrocephalus	**Sperm whale**
Family Kogiidae	
Kogia breviceps	Pygmy sperm whale
K. sima	Dwarf sperm whale
Family Ziphiidae	
Hyperoodon ampullatus	**Northern bottlenose whale**
M. bidens	**Sowerby's beaked whale**
M. densirostris	Blainville's beaked whale
M. europaeus	Gervais' beaked whale
M. grayi	Gray's beaked whale
M. mirus	True's beaked whale
Ziphius cavirostris	**Cuvier's beaked whale**
Family Monodontidae	
Delphinapterus leucas	White whale, beluga
Monodon monoceros	Narwhal
Family Delphinidae	
Delphinus delphis	**Short-beaked common dolphin**
Feresa attenuata	Pygmy killer whale
Globicephala macrorhynchus	Short-finned pilot whale
G. melas	**Long-finned pilot whale**
Grampus griseus	**Risso's dolphin**
Lagenodelphis hosei	Fraser's dolphin
Lagenorhynchus acutus	**Atlantic white-sided dolphin**
L. albirostris	**White-beaked dolphin**
Orcinus orca	**Killer whale**
Peponocephala electra	Melon-headed whale
Pseudorca crassidens	False killer whale
Stenella coeruleoalba	**Striped dolphin**
S. frontalis	Atlantic spotted dolphin
Tursiops truncatus	**Common bottlenose dolphin**
Family Phocoenidae (porpoises)	
Phocoena phocoena	**Harbour porpoise**

NOTE: Species in **bold** occur regularly in the ASCOBANS Agreement Area

APPENDIX 2: Status of Cetacean Species Occurring in NW Europe, by Country

							COUNTRY							
CETACEAN SPECIES	**NO**	**DK**	**SE**	**FI**	**PO**	**LI**	**DE**	**NL**	**BE**	**UK**	**IE**	**FR**	**ES**	**PT**
a) baleen whales and large toothed whales														
Bowhead whale	RAR[1]	-	-	-	-	-	-	-	-	VAG	-	-	-	-
N. Atlantic right whale	VAG	-*	-	-	-	-	-	VAG	-*	VAG	VAG	-*	VAG	VAG
Minke whale	COM	COM[2]	RAR	-*	-	-	RAR	RAR	VAG	COM[3]	REG[4]	REG	REG	COM
Sei whale	RAR	VAG	RAR	-	-	-	VAG	VAG	VAG	RAR	REG	RAR	RAR	REG
Bryde's whale	-	VAG	-	-	-	-	-	-	-	-	-	-	-	-
Blue whale	RAR	-*	-	-	-	-	-	-*	-*	RAR	RAR	VAG	RAR	VAG
Fin whale	REG	VAG	RAR	-	-	-	VAG	VAG	VAG	REG	REG	RAR	REG	REG
Humpback whale	COM	VAG	VAG	VAG	-*	-	VAG	VAG	-*	RAR	RAR	VAG	RAR	RAR
Sperm whale	REG	RAR	RAR	-	VAG	-	VAG	VAG	VAG	RAR	RAR	REG	REG	REG
b) small cetaceans														
Pygmy sperm whale	-	-	-	-	-	-	-	-*	-	VAG	VAG	RAR	VAG	VAG
Dwarf sperm whale	-	-	-	-	-	-	-	-	-	VAG	-	VAG	VAG	-
Northern bottlenose whale	REG	VAG	RAR	-	-*	-	VAG	VAG	-*	REG	REG	RAR	REG	-
Sowerby's beaked whale	RAR	VAG	RAR	-	-	-	VAG	VAG	-*	RAR	RAR	RAR	RAR	RAR
Blainville's beaked whale	-	-	-	-	-	-	-	VAG	-	VAG	-	VAG	VAG	VAG
Gervais' beaked whale	-	-	-	-	-	-	-	-	-	-	VAG	VAG	-	RAR
Gray's beaked whale	-	-	-	-	-	-	-	-*	-	-*	-	-*	-	-
True's beaked whale	-	-	-	-	-	-	-	-	-	-	VAG	VAG	VAG	-
Cuvier's beaked whale	-	-	RAR	-	-	-	-	VAG	VAG	RAR	RAR	REG	REG	RAR
Beluga	RAR	VAG[5]	RAR	VAG	-*	VAG	VAG	VAG	VAG	VAG	-	-	-	-
Narwhal	RAR	-	VAG	-	-	-	-	-*	-	-*	-	-	-	-
Short-beaked common dolphin	VAG	REG	RAR	VAG	VAG	-	VAG	RAR	VAG	COM	COM	COM	COM	COM
Pygmy killer whale	-	-	-	-	-	-	-	-	-	-	-	VAG	VAG	-
Short-finned pilot whale	-	-	-	-	-	-	-	-	-	-	-	VAG	VAG	-
Long-finned pilot whale	COM[6]	RAR	RAR	-	-	-	VAG	VAG	VAG	COM	COM	COM	COM	COM
Risso's dolphin	VAG	-*	RAR	-	-	-	VAG	VAG	RAR	REG	REG	REG	REG	COM
Fraser's dolphin	-	-	-	-	-	-	-	-	-	VAG	-	VAG	-	VAG
Atlantic white-sided dolphin COM	RAR	RAR	-	-	-	VAG	RAR	VAG	COM	COM	RAR	RAR	RAR	
White-beaked dolphin	COM	COM[7]	RAR	RAR	RAR	-	RAR	REG	RAR	COM	REG	RAR	VAG	-
Killer whale	REG	REG[8]	VAG	-	-	-	VAG	VAG	VAG	REG	REG	RAR	RAR	REG
Melon-headed whale	-	-	-	-	-	-	-	-	-*	-*	-	VAG	-	-
False killer whale	VAG	-*	-*	-	-	-	-*	-*	-	VAG	VAG	VAG	VAG	RAR
Striped dolphin	VAG	VAG	VAG	-	VAG	-	VAG	VAG	VAG	RAR	RAR	COM	COM	COM
Atlantic spotted dolphin	-	-	-	-	-	-	-	-	-	-	-	VAG	-	-
Bottlenose dolphin	VAG	VAG	VAG	-*	-	VAG	VAG	RAR	RAR	COM	COM	COM	COM	COM
Harbour porpoise	COM	COM	COM	RAR	RAR	VAG	COM	COM	COM	COM	COM	REG	COM	COM

NOTES

Countries: NO = Norway; DE = Denmark; SE = Sweden; FI = Finland; PL = Poland; LI = Lithuania; DE = Germany; NL = Netherlands; BE = Belgium, UK = United Kingdom; IE = Ireland; FR = (Atlantic) France; ES = (Atlantic) Spain (excl. Canaries); PT = (Atlantic) Portugal (excl. Azores and Madeira)

For Latvia and Estonia, there is insufficient information on status of most species, although no species is regular, and harbour porpoise occurs at best as a vagrant

Cetacean Status (based on records since 1980): VAG = Vagrant; RAR = Rare; REG = Regular (but Uncommon); COM = Common; - = Not Recorded; * = Record(s) before 1980

Despite frequent references to it in handbooks, rough-toothed dolphin, *Steno bredanensis*, has not been recorded with certainty from the ASCOBANS region. There are two nineteenth century records ascribed to *Steno* from the Netherlands, one based only upon a description and drawing, and the other on a skull found in a ditch, but no skeletal evidence of the former has been found, and the origins of the latter are uncertain and may derive from a sailor's travels elsewhere in the world.

[1] VAG in northern Norway only, [2] REG in Kattegat/Baltic, [3] but REG in Channel and Southern North Sea, [4] but COM in Southwest; [5] but annual, periodically,
[6] but periodic, at other times RAR, [7] REG in Kattegat/Baltic, [8] RAR in Kattegat/Baltic

Source: Waring *et al.* (2008)