

Background information on marine mammals for Strategic Environmental Assessment 8

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Non-technical summary

Distribution and abundance

There is extensive information on the distribution and abundance of grey and harbour seals around Britain from annual aerial surveys of breeding colonies and haul-out sites and a large body of data from satellite telemetry studies. Information on cetacean abundance and distribution comes from both dedicated and opportunistic sightings surveys, supported by voluntary organisations, industry, governmental and international agencies.

Four cetacean species occur frequently in the SEA8 area: minke whale, bottlenose dolphin, short-beaked common dolphin, and harbour porpoise. Long-finned pilot whale and Risso's dolphin are also regularly encountered. Grey seals are present in the SEA8 area but in low densities relative to the rest of UK shelf waters. Harbour seals are uncommon in the SEA8 area.

The SCANS-II survey for cetaceans in July 2005 covered the continental shelf waters of the SEA8 area. Abundance was estimated for four species. The harbour porpoise was the commonest cetacean in the region. In July 2005 there were an estimated 80,600 (CV = 0.50) porpoises in the Celtic Sea and adjacent waters and 40,900 (CV = 0.38) in the Channel and contiguous southern North Sea. The short-beaked common dolphin is also abundant with an estimated 11,100 (CV = 0.61) individuals in the Celtic Sea and contiguous shelf waters, and 14,300 (CV = 1.66) in the Channel and contiguous southern North Sea. Abundance in winter is not known but densities in the western Channel increase 10-fold during the wintertime.

Minke whale abundance was estimated as 1,200 (CV = 0.96) in the Channel and contiguous southern North Sea and 1,720 (CV = 0.43) in the Celtic Sea and adjacent waters. Bottlenose dolphin abundance was estimated as 395 (CV = 0.74) in the Channel and contiguous southern North Sea and 5,370 (CV = 0.49) in the Celtic Sea and adjacent shelf waters; these estimates incorporate resident populations off France totalling more than 200 individuals.

Ecological importance

The SEA8 area is ecologically important for harbour porpoise, common dolphin, bottlenose dolphin and minke whale because of their abundance there. However, there is little specific information on the diet of these species in this area. In the SEA8 area the common dolphin has been found to feed predominately on horse mackerel, sardines, mackerel, *Trisopterus* spp and gobies. Off southern England, harbour porpoise diet was dominated by whiting, poor cod and scad.

The abundance and availability of fish, especially those species mentioned above, is clearly of prime importance in determining the reproductive success or failure of marine mammals in this area, as elsewhere. Changes in the availability of principal forage fish may therefore be expected to result in population level changes of marine mammals. It is currently not possible to predict how any particular change in fish abundance would be likely to affect any of these marine mammal populations.

Sensitivity to disturbance, contamination and disease

Noise

Offshore oil and gas production is noisy. Each stage of the oil extraction process produces loud and potentially disturbing or even damaging sounds. Exploration entails seismic surveys that produce intense low frequency impulse noise, extraction includes drilling, increased vessel traffic, pipeline laying and seismic site surveys, and decommissioning can involve explosive removals.

There is an increasing awareness of the importance of sound to marine mammals. Any man-made noise could potentially have an effect on a marine mammal. The effects could range from mild irritation through impairment of foraging or disruption of social interactions to hearing loss and in extreme cases may lead to injury or even death. Most of the noise generated by offshore oil operations is low frequency, mostly <1kHz, although higher frequency sounds are also generated. Seals are known to be sensitive to

those frequencies whereas small (toothed) cetaceans are relatively insensitive to low frequencies. There are no direct measurements of either the frequency range or sensitivities of hearing in large whales, but circumstantial evidence suggests that they may have good low frequency hearing.

Seismic surveys have been shown to cause avoidance behaviour in grey and harbour seals, and in a range of large cetacean species. Current mitigation methods are probably generally effective in preventing physical damage. The development of 4D or time lapse seismic surveys means that areas with intense activity may be subjected to repeated disturbance. The effects of such repeated surveys are unknown, but minor or even insignificant transient effects may become important if disturbance is repeated/intensified.

There are no reliable data to suggest that vessel noise or drilling noise adversely affect seals or small cetaceans but there are indications that large whales may avoid areas of intense activity.

A relatively new source of noise in UK coastal waters is that associated with the construction and running of offshore wind farms. To date there is limited information on the noise generated during each of the survey, construction and operation phases. Harbour porpoises have shown equivocal responses to construction activity; harbour seals and harbour porpoises have shown relatively mild aversive behavioural responses to the playback of underwater noise from a simulated 2MW wind turbine.

Contaminants

A substantial amount of information is available on the uptake of lipophilic contaminants by marine mammals, such as polychlorinated biphenyls, DDTs and chlorinated pesticides. Other studies on captive and wild populations have shown that these compounds probably have toxic effects on the reproductive and immune systems. Certain heavy metals such as mercury, lead, cadmium, copper and zinc are taken up by marine mammals although there is little evidence that these cause substantial toxic responses, except at high concentrations. Cetacean species which feed lower down the food chain may be at risk from exposure to polyaromatic hydrocarbons, although very little is known about current exposure levels or the effects of chronic exposure in marine mammals.

Oil spills

Direct mortality as a result of contaminant exposure associated with major oil spills has been reported, e.g. following the Exxon Valdez oil spill in Alaska in 1989. Many animals exposed to oil developed pathological conditions including brain lesions. Additional pup mortality was reported in areas of heavy oil contamination compared to unoiled areas.

More generally, marine mammals are less vulnerable than seabirds to fouling by oil, but they are at risk from hydrocarbons and other chemicals that may evaporate from the surface of an oil slick at sea within the first few days. Symptoms from acute exposure to volatile hydrocarbons include irritation to the eyes and lungs, lethargy, poor coordination and difficulty with breathing. Individuals may then drown as a result of these symptoms.

Grey seals (there are few harbour seals in the SEA8 area) come ashore regularly throughout the year between foraging trips and additionally spend significantly more time ashore during the moulting period (February-April) and particularly the pupping season (September-December). Animals most at risk from oil coming ashore on seal haul-out sites and breeding colonies are neonatal pups, which are therefore more susceptible than adults to external oil contamination.

Oil dispersants

There have been no specific studies on the direct acute or chronic toxicity of oil dispersants to seals and cetaceans.

Disease

A small-scale survey of anthropogenic bacteria, including *Salmonella* and *Campylobacter*, has been conducted in seals but there is no information on the occurrence of anthropogenic viruses, such as enteroviruses.

Bycatch and other non-oil related management issues

Bycatch

The accidental capture (bycatch) of marine mammals in fishing gear is an issue of current concern throughout EU waters and beyond. Bycatch levels in the SEA8 area are relatively high compared with

other areas around the UK, due to the presence of large amounts of gillnetting, significant levels of pelagic trawling during the winter and high densities of harbour porpoise and common dolphin. The number of porpoises currently caught in static nets in the region annually is unknown but is likely in the hundreds at least. In the western channel common dolphins seem especially vulnerable to bycatch in pelagic pair trawls targeting bass in the winter months, when common dolphin densities are at a peak. Total mortalities in UK bass pair teams peaked at over 400 animals in the 2003-2004 winter but have since declined to less than 100 in 2005-2006.

Ship collisions

A potential source of mortality to cetaceans in this and other areas is through collisions with shipping. In other areas, where ships are numerous and cetacean numbers are depleted, this is a serious cause for concern. The frequency of such events in the SEA8 area is unknown and consequently this has not been identified as a significant source of additional mortality in this region.

Conservation frameworks

Marine mammals are included in a wide range of conservation legislation. All species are listed on Annex IV (Animal and Plant Species of Community Interest in Need of Strict Protection) of the European Commission's Habitats Directive. Under Annex IV, the keeping, sale or exchange of such species is banned as well as deliberate capture, killing or disturbance. The harbour porpoise, bottlenose dolphin, grey seal and harbour seal are also listed in Annex II of the Habitats Directive. Member countries of the EU are required to consider the establishment of Special Areas of Conservation (SACs) for Annex II species. SACs have been established for the bottlenose dolphin in the Moray Firth (one) and in Cardigan Bay (two). No SACs have yet been established for the harbour porpoise. A number of terrestrial SACs have been established for grey and harbour seals around the coast of the UK. There are currently no marine SACs for seals.

Under the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS) provision is made for protection of specific areas, monitoring, research, information exchange, pollution control and heightening public awareness. Measures cover the monitoring of fisheries interactions and disturbance, resolutions for the reduction of by-catches in fishing operations, and recommendations for the establishment of specific protected areas for cetaceans.

In British waters, all species of cetacean are protected under the Wildlife and Countryside Act 1981 and the Wildlife (Northern Ireland) Order 1985. Whaling is illegal under the Fisheries Act 1981. Guidelines to minimise the effects of acoustic disturbance from seismic surveys, agreed with the oil and gas industry, were published by the Department of the Environment in 1995 and are revised from time to time. In 1999, the Department of the Environment, Transport and the Regions produced two sets of guidelines aimed at minimising disturbance to cetaceans. Grey and harbour seals in the vicinity of fishing nets can be killed to prevent damage to the nets or to fish in the nets under the Conservation of Seals Act 1970. Both species are protected during the breeding season; however, licences to kill seals may be granted for any time of the year for specific listed purposes.

Conclusions

- The SEA8 area is an important area for some cetacean species. Harbour porpoises are abundant in summer at least, and common dolphins are abundant year round and especially in winter. Bottlenose dolphins and minke whales are relatively common in summer at least. Long-finned pilot whales and Risso's dolphins also occur regularly.
- These species are thus important predators in this region although there is little specific information on their diet. Because of the link between the abundance and availability of fish prey and the reproductive success of marine mammals, changes in the availability of principal forage fish may be expected to result in population level changes of marine mammals. It is currently not possible to predict the extent of this.
- Seals are sensitive to the low frequency sounds generated by oil exploration and production but seal densities are low in the SEA8 area. Small cetaceans are relatively insensitive to low frequencies. Circumstantial evidence suggests that large whales may have good low frequency hearing. There are no reliable data to suggest that vessel noise or drilling noise adversely affects seals or small cetaceans.

- A relatively new source of noise in UK coastal waters is that associated with the construction and running of offshore wind farms. To date there is limited information on the noise generated during each of the survey, construction and operation phases. Harbour porpoises have shown equivocal responses to construction activity; harbour seals and harbour porpoises have shown relatively mild aversive behavioural responses to the playback of underwater noise from a simulated 2MW wind turbine.
- Contaminants, such as polychlorinated biphenyls, DDTs and chlorinated pesticides probably have toxic effects on the reproductive and immune systems of marine mammals. There is little evidence that heavy metals cause substantial toxic responses, except at high concentrations. Cetacean species which feed lower down the food chain may be at risk from exposure to polycyclic aromatic hydrocarbons, although very little is known about current exposure levels or the effects of chronic exposure in marine mammals.
- Major oil spills are likely to result in direct mortality. More generally, marine mammals are less vulnerable than seabirds to fouling by oil, but they are at risk from chemicals evaporating from the surface of an oil slick at sea within the first few days. Individuals may drown as a result of associated symptoms. Neonatal seal pups are at risk from oil coming ashore.
- Bycatch levels in the SEA8 area are relatively high compared with other SEA areas, due to the presence of large amounts of gillnetting, significant levels of pelagic trawling during the winter and high densities of harbour porpoise and common dolphin.

1. DISTRIBUTION AND ABUNDANCE

1.1 Introduction

This section summarises information on the distribution and abundance of marine mammals occurring in waters off the south west coast of the United Kingdom, with particular reference to the SEA8 area. The SEA8 area encompasses the English Channel, Celtic Sea and the western Approaches, an area that is very important to a number of cetaceans, especially during the wintertime. This area is characterised by the presence of hydrological fronts creating important primary and secondary productivity (Kiszka *et al.*, 2004, and ref. therein). The western English Channel is in a boundary area between oceanic and neritic waters, and is influenced by strong tides and freshwater inputs, which create well-mixed waters.

Over the last hundred years, a considerable period of change in the environment of the English Channel has been documented, with significant periods of warming (1921-1961; 1985-present) and cooling (1962-1980) which resulted in the abundance of key species undergoing dramatic shifts in distribution (Southward *et al.*, 2005). During the warm water period of 1921-1961 (Russell cycle), a gradual change occurred in the composition of plankton and fish species in the English Channel, which resulted in an increase in the presence of warm-water species, especially from 1936-1965 (Southward *et al.* 2005, and ref. therein). This may have led to changes in the marine mammal community in the English Channel and contiguous waters at that time, as a decline was noted in strandings of common dolphins along the southwest coast of the UK and Ireland (Evans and Scanlan, 1989; Murphy, 2004; Murphy *et al.*, 2006). Since 1965 however, many of the conditions prevailing in the 1920s returned, along with an increase in strandings of common dolphins along the southwest coast of England (Evans and Scanlan 1989) and the southern and western coasts of Ireland (Murphy 2004). In the future however, variations in the abundance and distribution of marine mammals in the SEA8 area could alter again, due to changes in climate, i.e. North Atlantic Oscillation.

The SCANS (Small Cetacean Abundance in the North Sea and adjacent waters) and SCANS-II (Small cetaceans in the European Atlantic and North Sea) surveys took place in July 1994 and July 2005, respectively (Hammond *et al.*, 2002; Hammond *et al.* 2008). Both surveys covered the entire continental shelf zone of the SEA8 area and are presented in the following sections.

Other effort-related data from Sea Watch Foundation and the Joint Nature Conservation Committee's Seabirds at Sea Team are also included (Reid *et al.* 2003).

Ten species have been recorded in the region. Of these, six species are known to occur regularly: harbour porpoise, bottlenose dolphin, short-beaked common dolphin, Risso's dolphin, long-finned pilot whale, and minke whale. Four further species are infrequently recorded: killer whale, striped dolphin, white-sided dolphin and white-beaked dolphin, and there are rare sightings of a further two species: sei whale and fin whale. In the following sections, each of the more abundant species is briefly described with particular reference to its distribution and abundance in the SEA8 area.

1.2 Baleen whales

1.2.1 Minke whale (*Balaenoptera acutorostrata*)

The minke whale is widely distributed in all the major oceans of the world from tropical to polar seas, though it is most abundant in relatively cool waters, and on the continental shelf (in depths of 200 m or less). In the Northeast Atlantic, it ranges from Norway to France, and into the northern and central North Sea. Within UK waters, it is most frequently sighted in the north-western North Sea and the Hebrides, and it is rare in the southernmost North Sea and eastern half of the English Channel. In the western English Channel it is evenly distributed in low numbers along the continental shelf edge, and is almost absent from the deeper parts of the Bay of Biscay, although concentrations of sightings around the Brittany coast and the northern edge of the Bay of Biscay have been reported (Reid *et al.* 2003). The species has been recorded in every month of the year in UK waters, but is mainly sighted near the coast, between May and September (Evans, 1990; Northridge *et al.*, 1995; Evans *et al.*, 2003). Figure 1 shows the distribution of minke whales in the SEA8 area over all seasons.

The SCANS-II survey estimated 1,200 (CV = 0.96) minke whales in the English Channel and contiguous southern North Sea, and 1,720 (CV = 0.43) individuals in the Celtic Sea and Bay of Biscay shelf waters in July 2005. A total of 18,600 (CV = 0.30) animals were observed in western European continental shelf waters during July 2005 (Hammond *et al.* 2008). Density surface models predicted highest

concentrations of animals in the central North Sea, off Norway, north-east Scotland, southern Ireland and south-west England.

Minke whales in the SEA8 area are considered by the International Whaling Commission as part of a single Northeast Atlantic stock, however some population differentiation within this region may exist between the North Sea and the rest of the northeast Atlantic (Andersen *et al.*, 2003). There are about 150,000 minke whales in the north-eastern and central North Atlantic (<http://www.iwcoffice.org/index.htm>).

1.3 Toothed whales

1.3.1 Common bottlenose dolphin (*Tursiops truncatus*)

The bottlenose dolphin has a worldwide distribution in tropical and temperate seas of both Hemispheres. Along the Atlantic seaboard of Europe, the species is frequently reported in neritic waters off Spain, Portugal, north-west France, southern and western Ireland, north-east and south-west Scotland, in the Irish Sea, and in the English Channel (Evans *et al.*, 2003). Data obtained during the SCANS 1994 survey indicated that the western Celtic Sea is a relatively important area for bottlenose dolphins (Hammond *et al.*, 1995). Reid *et al.*, (2003) also reported large aggregations of bottlenose dolphins off southwest of Ireland, in the vicinity of the shelf break, southwards towards the French coast, and also off western Ireland. In coastal waters, bottlenose dolphins favour river estuaries, headlands and sandbanks, mainly where there is uneven bottom relief and/or strong tidal currents (Lewis and Evans, 1993; Wilson *et al.*, 1997; Liret, 2001; Ingram and Rogan, 2002). In UK coastal waters, peak sightings occur between July and October, with a secondary peak reported in some localities during March and April (Evans, 1992a). However, animals are present near to shore in every month of the year (Evans, 1992). Figure 2 shows the distribution of bottlenose dolphins in the SEA8 area over all seasons.

Overall sightings rates in the SEA8 are lower than those reported in the SEA6 area and adjacent waters along the shelf break, west of France, and off the southwest coast of Ireland. The SCANS-II survey estimated 395 (CV = 0.74) bottlenose dolphins in the English Channel and contiguous southern North Sea, and 5,370 (CV = 0.49) individuals in the Celtic Sea and adjacent shelf waters in July 2005. A total of 12,600 (CV = 0.27) animals were estimated in western European continental shelf waters during July 2005 (Hammond *et al.* 2008).

In the English Channel and off the northwest coast of France 85 individuals have been reported (Liret *et al.*, 1998). Off the coast of France, the majority of individuals are concentrated around the Cotentin peninsula and in the Mont-Saint-Michel Bay, with group sizes ranging between 1-30 (average 11) individuals (Kiszka *et al.* 2004). Two important groups of resident bottlenose dolphins have been identified off the French Brittany coast, around Ile de Sein, and the Molene archipelago, consisting of 17 and 35-50 individuals respectively (Liret, 2001; Kiszka *et al.* 2004, and ref. therein). Off Normandy, Kiszka *et al.* (2004) suggested that the resident population consist of approximately 80 individuals.

A population of approximately 213 individuals (95% CI = 183-279) has been reported for the Cardigan Bay area (Baines *et al.*, 2002). The population is not closed as individuals may join up for periods of time from elsewhere (Reid *et al.* 2003), and sightings of individuals initially reported off the south-west coast of England, have been observed in Welsh waters (Wood, 1998).

Williams *et al.* (1996) reported seasonal movements in the English Channel, with the majority of sightings reported off the Cornish coast during the wintertime, followed by an eastwards movement during the spring, to waters as far as the east Sussex coast. During the summertime highest sightings were reported from Lyme Bay eastwards, and in the autumn the majority of sightings were reported off the Dorset coast, east of the Isle of Wight. The seasonal movements of bottlenose dolphins in this area have been linked to the distribution of both fish and chlorophyll (Sykes *et al.*, 2003). In contrast, Wood (Wood, 1998) suggested a northerly shift in distribution of bottlenose dolphins off the Cornish coast across the Bristol Channel into Welsh waters. Overall, it has been suggested that animals along the French Channel coast form very stable groups that are resident in small areas, whereas those along the southern English coast are wider-ranging (Reid *et al.* 2003, and ref therein).

1.3.2 Short-beaked common dolphin (*Delphinus delphis*)

The short-beaked common dolphin has a worldwide distribution and inhabits both oceanic and shelf-edge waters of tropical, subtropical and temperate seas of the Atlantic and Pacific Oceans. It is abundant and

widely distributed throughout the Northeast Atlantic, with summer sightings as far north as approx. 70°N latitude, west of Norway (Murphy 2004, and ref. therein, Cañadas *et al.* in press, and ref. therein), although the majority of common dolphin sightings have been reported in waters south of 60°N. Strong seasonal shifts in their distribution in the Northeast Atlantic have been noted, with wintertime movements onto the Celtic Shelf and into the western English Channel (Northridge *et al.*, 2004), and at this time, high densities have been reported throughout the SEA8 area. The mating/calving period for this species in the Northeast Atlantic extends from May to September (Murphy *et al.*, 2005; Murphy and Rogan, 2006); at this time the majority of sightings have been reported along and off the continental shelf slope. Overall, relatively few sightings have been reported in the eastern English Channel and the North Sea. The majority of reported UK common dolphin strandings are along the southwest coast (Sabin *et al.*, 2002). Figure 3 shows the distribution of common dolphins in the SEA8 area over all seasons.

Common dolphins are found in a wide range of group sizes from small schools to large concentrations of 1000 to 5000 individuals (Murphy, 2004, and ref. therein). In the SEA8 area, Kiszka *et al.* (2004) reported group sizes of 2-500 individuals (average group size 25 individuals) off the French Channel coast, with the largest groups observed south of Guernsey.

In July 2005, there were an estimated 11,100 (CV = 0.61) individuals in the Celtic Sea and contiguous shelf waters, and 14,300 (CV = 1.66) in the Channel and adjacent southern North Sea. Abundance in winter estimate is not known but densities in the western Channel increase 10-fold during the wintertime (Brereton *et al.*, 2005).

To date, low levels of population differentiation have been reported in the Northeast Atlantic, using samples obtained from waters off Portugal to Scotland (Natoli *et al.* 2006; Murphy *et al.* 2006; Amaral *et al.* 2007), which may be attributed to common dolphins off Portugal mixing with the Mediterranean Sea population, or animals further south of this sampled area (Murphy *et al.* 2006). Viricel (2006) did not find any significant variation in mtDNA and microsatellites between samples obtained from oceanic and neritic waters of the Bay of Biscay and the English Channel. Mirimin *et al.* (2005) also reported a lack of significant variation between *D. delphis* inhabiting the English Channel, the Celtic Sea and off the Irish Coast, using mtDNA and microsatellites, which suggests that one population inhabits this area.

1.3.3 Long-finned pilot whale (*Globicephala melas*)

The long-finned pilot whale has a worldwide distribution in temperate and sub-polar seas of both Hemispheres. It is common and widely distributed in deep North Atlantic waters. In British and Irish waters, long-finned pilot whales occur mainly along the continental shelf slope, particularly around the 1,000 metre isobath. In the SEA8 area, they are predominately sighted in the western English Channel (Figure 4) off the south-west coast of England, during the autumn and early spring (Evans, 1980). Along the French side of the Channel, long-finned pilot whales were sighted primarily in north-eastern Normandy and around the Channel Islands, throughout the year (Kiszka *et al.* 2004). Group sizes of pilot whales off the French Channel coast were highly variable, ranging from 1-150 animals, with an average group size of 17 individuals (Kiszka *et al.* 2004). The majority of reported UK pilot whale strandings are along the southwest coast (Sabin *et al.*, 2002).

Pilot whales have a propensity to mass live strand, due to the cohesive nature of their social groups. Pilot whales are hunted by the Faroese; there are an estimated 780,000 animals in the central and Northeast Atlantic (<http://www.iwcoffice.org/index.htm>). There are no abundance estimates for this species in the SEA8 area.

1.3.4 Risso's Dolphin (*Grampus griseus*)

Risso's dolphin is widely distributed in tropical and temperate seas of both Hemispheres, occurring in small numbers along the Atlantic European seaboard from the Northern Isles south to northwest France, the southern Bay of Biscay, around the Iberian Peninsula and east into the Mediterranean Sea. In UK waters, the majority of sightings have been reported around the Hebrides, but the species is regularly seen around the Northern Isles, in the southern Irish Sea and off southwest Ireland. It is rare in the North Sea and all but the western end of the Channel. In French Channel waters, the majority of sightings were reported off the northern Brittany coast, with group sizes ranging from 1-8 individuals (Kiszka *et al.* 2004). No sightings were reported in the eastern Channel (Kiszka *et al.* 2004). There are no abundance estimates for this species in the SEA8 area.

1.3.5 Harbour porpoise (*Phocoena phocoena*)

The distribution of the harbour porpoise is restricted to temperate and sub-arctic (mainly 5-14° C) seas of the Northern Hemisphere. In the Northeast Atlantic, it is common and widely distributed on the continental shelf (mainly at depths of 20-200 m) from the Barents Sea and Iceland, south to the waters off the Iberian coast. It is the most frequently observed (and stranded) cetacean in British and Irish waters.

During the SCANS survey in July 1994, harbour porpoises were not sighted in the Channel or the southern North Sea but there were an estimated 36,280 (CV = 0.57) animals in the Celtic Sea and adjacent shelf waters (Hammond *et al.* 2002). In July 2005, there were an estimated 80,600 (CV = 0.50) porpoises in the Celtic Sea and adjacent waters and 40,900 (CV = 0.38) in the Channel and contiguous southern North Sea.

In recent years, there has been a notable increase of individuals off French, Belgian and Dutch coasts in the Channel and southern North Sea (Camphuysen, 2004; Kiszka *et al.*, 2004), especially during the winter and early spring. There has also been a significant increase in the number of harbour porpoise reported stranded along the southwest coast of the UK in recent years, especially since 2000, during January and April (Jepson *et al.* 2005). Figure 5 shows the distribution of harbour porpoise throughout the year in the SEA8 area. In this area, the majority of individuals were sighted off the south coast of Wales and west of Cornwall.

Metrical studies using skeletal material, along with studies of tooth ultra-structure and genetics together suggest that populations of harbour porpoises may exist in the North Sea and adjacent waters, with possible separate populations occurring in the Irish Sea (Duke 2003, Thatcher 2005), off the south west coast of the UK (Thatcher, 2005), British and Danish North Sea, with movement of animals from both North Sea populations southwards, to waters off the Netherlands (Walton, 1997; Andersen, 2003). It has also been reported that harbour porpoises off the UK southwest coast population are significantly larger than animals that strand along the Welsh coastline (Jepson, 2003). It is not known however if individuals from this UK southwest coast population mix with animals off France. Genetic evidence also indicates that males disperse more widely than females (Walton, 1997; Tolley *et al.*, 1999; Andersen *et al.*, 2001). The mating/calving periods for the harbour porpoise ranges from May to August in the Northeast Atlantic (Learmonth, 2006).

1.3.6 Other species occasionally sighted

Several other species of toothed cetacean have been recorded in the SEA8 area in low numbers: killer whale, striped dolphin, white-sided and white-beaked dolphin. For beaked whales, only a handful of strandings have been recorded in the area. Fin and sei whales are occasionally seen. There are no abundance estimates for these species in the SEA8 area.

Killer whale (Orcinus orca)

The killer whale has a worldwide distribution in tropical, temperate and polar seas in both Hemispheres (with greatest abundance at higher latitudes). It is widely distributed on the Atlantic seaboard of northern Europe, mainly around Iceland (Sigurjónsson *et al.*, 1988), western Norway (Similä *et al.*, 1996), and northern Scotland (Luque *et al.* 2006). Killer whales are also sighted off the Iberian Peninsula and in the eastern Mediterranean Sea (Notarabartolo-di-Sciara, 1987)

Striped dolphin (Stenella coeruleoabla)

In the last twenty years, a northwards movement in the distribution of striped dolphin (normally classified as a warm water species) has been seen. Striped dolphins were documented in UK stranding records between 1923 and 1939, but were not reported stranded again until 1975. Following this, they have been reported in almost every year, with increasing frequency. The distribution of the striped dolphin is normally restricted to deep offshore waters of 1,000m or more, although they are reported stranded along both the UK and the French Channel coasts (Sabin *et al.*, 2002; Kiszka *et al.*, 2004).

White-beaked dolphin (Lagenorhynchus albirostris)

White-beaked dolphins are restricted to the North Atlantic. In the eastern North Atlantic their range extends from the British Isles to Spitsbergen. They are mainly distributed over the continental shelf, and in the North Sea and adjacent areas (Northridge *et al.* 1995; Hammond *et al.* 2008).

Atlantic white-sided dolphin (Lagenorhynchus acutus)

Atlantic white-sided dolphins are confined to the North Atlantic. They share most of their range with the white-beaked dolphin, but in the eastern North Atlantic they adopt a mainly offshore distribution. At sea, the two species can be difficult to distinguish and there is a tendency for them to be recorded simply as *Lagenorhynchus spp.* Around Britain, Atlantic white-sided dolphins have been recorded mainly off the north and west coasts.

1.4 Pinnipeds

1.4.1 Grey seal (*Halichoerus grypus*)

Grey seals are restricted to the North Atlantic and adjacent seas. There are three recognised populations: the northwest Atlantic (breeding primarily on Sable Island, Canada and in the Gulf of St Lawrence); the Baltic Sea; and the northeast Atlantic (breeding primarily on offshore islands around the British Isles but also in Iceland, the Faroe Islands, France, the Netherlands, central and northern Norway, and around the Kola peninsula in Russia). Grey seals haul out on land between foraging trips and for pupping and moulting, when they can form large colonies or aggregations. Timing of pupping differs throughout the range of the species. In the SEA8 area it occurs relatively earlier (September/October) than in northern Britain. Moulting occurs February - April.

The British grey seal population is currently estimated at around 133,000 (95% confidence interval: 107,000-171,000) individuals (SCOS, 2007), based on data from breeding colonies. The distribution of grey seals at haul-out sites around Britain in summer is shown in Figure 6. There are no estimates for the SEA8 area but Figure 6 shows that the southwest of Britain is a low density area relative to the rest of the UK.

The distribution of grey seals at sea in the SEA8 area has been studied by tracking animals fitted with satellite relay data loggers (Matthiopoulos *et al.* 2004; SMRU, unpublished data). Figure 7 shows the predicted area usage by grey seals tagged through 2002 based on these data, including animals tagged off Brittany, France, and counts of animals at haul-out sites. Data collected in the Irish Sea in 2004 are not included in this map; Figure 8 shows the predicted area usage of animals tagged at haul-out sites in Wales. Figures 7 and 8 show that usage of the SEA8 area by grey seals is very low.

1.4.2 Harbour (or common) seal (*Phoca vitulina*)

The harbour seal is one of the most widespread pinniped species and has a practically circumpolar distribution in the Northern Hemisphere. However, there are very few harbour seals in the SEA8 area. The distribution of harbour seals at haul-out sites around Britain in summer is shown in Figure 9.

2. ECOLOGICAL IMPORTANCE

The abundance and availability of fish, especially those species mentioned below, is clearly of prime importance in determining the reproductive success of marine mammals in this area, as elsewhere. Changes in the availability of principal forage fish may therefore be expected to result in population level changes of marine mammals. It is currently not possible to predict how any particular change in fish abundance would be likely to affect any of these marine mammal populations. However, as mentioned earlier, changes in plankton and fish communities in the English Channel between the 1930s and 1960s may have altered the distribution and/or density of marine mammals at that time.

2.1 Cetaceans

The six most frequently observed species of cetacean in the SEA8 area are the minke whale, bottlenose dolphin, short-beaked common dolphin, Risso's dolphin, long-finned pilot whale and harbour porpoise.

2.1.1 Minke whale

Minke whales are known to feed on a variety of fish species. Stephenson (1951) reported that most minke whales taken by commercial whaling in the UK waters of the North Sea during 1948 had been feeding on herring, with some mackerel and sand eels also reported. A more recent study by Olsen and Holst (2001) reported that in the North Sea, sandeels contributed 86.7% to the diet by weight, followed by mackerel (9.3%), whiting (2.4%), herring (1.1%) and Norway pout (0.5%). Whereas in the Norwegian

Sea the diet was dominated by Norwegian spring-spawning herring (100%), which is consistent with the large and dominant abundance of herring in the area (Olsen and Holst, 2001). The domination of pelagic species in the diet strongly indicates pelagic feeding behaviour in both areas (Olsen and Holst, 2001). De Pierrepont *et al.* (2005) identified poor cod, goby, cod and pollock in the stomach of one whale that stranded on the French Atlantic coast. There is no specific information on diet in the SEA8 area.

2.1.2 Common bottlenose dolphin

Analysis of the prey remains in ten stomachs from animals that were stranded and by-caught around Scotland, between 1990 and 1999, revealed that cod, saithe, and whiting were the main prey consumed, although several other fish species were also found, including salmon and haddock, as well as some cephalopods (Santos *et al.*, 2001). In French neritic waters, the predominant prey species reported in the stomachs of bottlenose dolphins were blue whiting, hake, scad, *Trisopterus* species, horse mackerel and *Loligo vulgaris* (Learmonth *et al.*, 2004; De Pierrepont *et al.*, 2005), and in Irish waters, haddock, saithe and pollock were the dominant prey species reported (Learmonth *et al.*, 2004). There is no specific information on diet in the SEA8 area.

2.1.3 Short-beaked common dolphin

The diet of common dolphin includes a variety of fish and squid. Prey remains identified in the stomachs of stranded specimens examined from UK and Irish waters included *Trisopterus* spp., mackerel, sardine, whiting, herring, sprat and sandeel. (Santos, 1998; Gosselin, 2001; Learmonth *et al.*, 2004). Horse mackerel, mackerel, Norway pout and sardines dominated the diet. Cephalopods included mainly *Loligo* spp., *Alloteuthis subulata*, *Ancistroteuthis lichtensteini*, *Todarodes sagittatus*, *T. eblane* and *Sepiolo atlantica*, but various other species of squid, octopus and cuttlefish were also consumed. In the SEA8 area the common dolphin predominately feeds on horse mackerel, sardines and mackerel (Kuiken *et al.*, 1994; Merrett *et al.*, 1995; Gosselin, 2001).

De Pierrepont *et al.* (2005) reported *Trisopterus* spp, and gobies as the main prey species consumed in French Channel waters. In French inshore waters, four taxa contributed to the majority of prey remains, anchovy, sardine, horse mackerel, and *Trisopterus* spp. Meynier (2004). Strong interannual and seasonal variations in the diet were found, reflecting prey availability in the area. Common dolphins caught in Irish and French tuna driftnets during the 1990s along and off the continental shelf slope in the summertime, were predominately feeding nocturnally, when the migrating deep scattering layer approaches the surface, on meso-pelagic fishes such as myctophids and squids (Hassani *et al.* 1997, Brophy 2003, Pusineri *et al.* 2007).

2.1.4 Risso's dolphin

There is a lack of dietary data for Risso's dolphin but they are generally assumed to restrict their feeding to squids.

2.1.5 Long-finned pilot whale

Long-finned pilot whales are predominantly squid feeders (Desportes and Mouritsen 1993). Stomach analysis of animals from the North Atlantic suggests that they also supplement their diet with small amounts of fish such as saithe, mackerel and blue whiting (Gannon *et al.*, 1997). In French Channel waters cuttlefish, mainly *Sepia* species, were the predominant prey consumed (De Pierrepont *et al.*, 2005). There is no other indication of diet for long-finned pilot whales within the SEA8 area.

2.1.6 Harbour porpoise

Off southern England, whiting was the dominant prey species consumed by harbour porpoise, followed by poor cod and scad (Roberts, 2005). Off Wales, whiting was also the dominant prey species consumed, followed by gobies (Roberts, 2005). In a study of diet of harbour porpoises from Scottish waters, whiting, sandeels, haddock/saithe/pollack and Norway pout/poor cod dominated (Santos *et al.*, 2004).

The harbour porpoise is the most abundance marine mammal species in the SEA8 area and adjacent waters. The significance of this species' predation from an ecological perspective has not been assessed.

2.2 Pinnipeds

Grey seals are not abundant and harbour seals are rare in the SEA8 area. Figures 7 and 8 show that the estimated at-sea usage of grey seals in the area is low. It is clear that the SEA8 area is not important foraging habitat for seals in comparison with the rest of UK waters.

3. SENSITIVITY TO DISTURBANCE, CONTAMINATION AND DISEASE

3.1 Noise

Marine mammals spend most, or all, of their lives at sea, and for the majority of that time they are submerged. Light is absorbed quickly in salt water and in many marine habitats visibility will be restricted to a few metres: thus vision may be of limited use for long range sensing. Sound, however, propagates efficiently through water and marine mammals use both active and passive acoustics for a variety of purposes e.g. finding prey, detecting predators, communication -often over great ranges- and probably navigation.

Many human activities generate sound in the water, e.g. shipping, ice breaking, oil and gas exploration, sonar and explosions, and some of these sounds are extremely intense. Often anthropogenic noise is in the low to mid frequency bands that propagate well and as a consequence anthropogenic noise can be detectable at substantial ranges. Recent technological developments have introduced many new sources of noise in offshore waters. For example, shipping is the dominant noise source at low frequencies in most locations yet this sound source was completely absent before the introduction of mechanised shipping. Ross (1976) estimated that shipping had caused levels of ambient noise to rise by 10dB between 1950 and 1975 and he predicted a rise of another 5dB by the end of the 20th Century. Such perturbation of the acoustic environment may have profound implications for marine mammals that evolved to function efficiently in a very different, rather quieter, ocean.

A relatively new source of noise in many UK coastal waters is that associated with the construction and running of offshore wind farms, which will be mainly restricted to shallow waters. There are proposals to develop additional wind farms in the SEA 8 area. Underwater noise is generated during each phase of the lifecycle of an offshore wind farm: during survey, construction and operation.

3.1.1 Effects of man-made sounds on marine mammals

Any anthropogenic noise could potentially affect a marine mammal if the sound signal falls within an animal's audible range. Effects could range from mild irritation, impairment of foraging or disruption of social interactions through to hearing loss and in extreme cases physical injury or even death.

Richardson et al. (1995) defined a series of zones of noise influence based on the ranges within which certain acoustic effects can be expected. They recognised four zones, three of which are generally thought of as occur at increasing sound level: (1) the zone of audibility (the area within which an animal is able to detect sound); (2) zone of responsiveness (the area in which the animal responds behaviourally or physiologically); (3) the zone of hearing loss (the area in which the received sound level is high enough to cause tissue damage resulting in either temporary threshold shift (TTS), permanent threshold shift (PTS) and (4) the zone of masking the region within which noise is strong enough to interfere with detection of other sounds, such as communication signals or echolocation clicks.

The extent of the zone of masking depends on the characteristics of sounds that might be masked as well as that of the noise itself. If the detection of very faint sounds is considered then the zone of masking could be almost as great as the zone of audibility. Recent research suggests that disruption of normal diving behaviour, which may be noise induced, could lead cetaceans to develop decompression sickness (e.g. Jepson et al. 2003; Fernandez et al. 2005). This would suggest that, in some cases, noise emissions at much lower sounds levels, such as within the zone of responsiveness (level 2), could potentially cause the same level of severe physiological trauma as those required for direct physical effects (level 4). Another general shortcoming of the "zone of influence" scheme is that it doesn't consider the temporal dimension. Some effects, such as TTS, correlate well with total acoustic energy exposure rather than received levels and the length of time that animals are exposed to a sound level is an important factor. Finally, it must be remembered that marine mammals operate in a three dimensional environment and zones shouldn't be considered as purely two dimensional. Notwithstanding these caveats, "Zones of Influence" have been a

widely used and influential framework for thinking about this issue and we use them to structure the following brief review.

3.1.1.1 Zone of audibility

This zone is defined by the range at which an animal can just detect the sound. For a sound of a particular frequency to be detected it must be both above the animal's absolute hearing threshold at that frequency and be detectable against the background noise level in that frequency band.

Both conditioned behavioural responses to sound playback and electrophysiological measurements of brain function have been used to measure hearing sensitivities for a number of marine mammal species (see Richardson *et al.* 1995 for a general overview and Nedwell, Edwards, Turnpenny and Gordon, 2004 for a comprehensive collation of available information.). Such research has thus far largely been confined to pinnipeds and small odontocetes that can be maintained in captivity. The resulting audiograms are typically U shaped with sensitivities declining steeply at high and low frequencies. Absolute sensitivity and frequency ranges vary markedly between marine mammal species and also between individuals.

Information on the hearing sensitivity of those species likely to be encountered in the SEA 8 area is summarised below

Hearing sensitivity of pinnipeds

Underwater audiograms have been measured for a range of phocid species and all show a similar pattern (Richardson *et al.* 1995). The audiograms for harbour seals are typical, indicating a fairly flat frequency response between 0.1 and about 40kHz, with hearing thresholds between 60 and 85 dB re 1 μ Pa. Sensitivity decreases rapidly at higher frequencies, but in the one animal tested at low frequency, the threshold at 0.1 kHz was 96 dB re 1 μ Pa. indicating good low frequency hearing (Table 1). No behavioural audiograms are available for the grey seal, but electro-physiological audiograms (based on auditory evoked potentials) showed a typical pinniped pattern over the range of frequencies tested (Ridgeway and Joyce, 1975). The fact that grey seals make low frequency calls also suggests that they also have good low frequency hearing (Table 2). Pinnipeds appear to be considerably less sensitive than humans to airborne sounds below 10 kHz.

Table 1. Hearing sensitivity of the harbour seal from underwater audiograms (Richardson *et al.* 1995).

Species	Low Freq. (kHz)	Threshold (dB re 1 μ Pa)	Best Freq. (kHz)	Threshold (dB re 1 μ Pa)	Upper Freq. (kHz)	Threshold (dB re 1 μ Pa)
Harbour seal	0.1	96	10-30	60-85	180	130

Table 2. Characteristic frequencies of vocalisations produced by grey seals.

Species	Frequency range of vocalisations (kHz)
Grey seal	0.1 – 3

Table 3. Hearing sensitivity of pinnipeds from in-air audiograms (Richardson *et al.* 1995).

Species	Lower Frequency (kHz)	Threshold (dB re 1 μ Pa)	Upper Frequency (kHz)	Threshold (dB re 1 μ Pa)
Harbour seal	0.1	95	20	85

Hearing sensitivity of baleen whales

There are no published audiograms for baleen whales. It is assumed that they are sensitive to sound of low and medium frequencies because they predominantly emit low frequency sounds, primarily at frequencies below 1 kHz with vocalisations of some species being largely infrasonic (<20Hz) sounds. Baleen whales react behaviourally to low frequency calls from conspecifics. However, these observations do not provide accurate indications of hearing thresholds.

Summary information on the frequency range of vocalisations of those species present in the SEA 8 area is shown in Table 4. The high upper frequencies quoted here are often outliers that may not be representative. Most baleen whale sounds are concentrated at frequencies less than 1 kHz, but sounds up to 8 kHz are not uncommon. The dominant call from fin whales is an infrasonic 20Hz pulse and in many oceans their calls are a prominent feature of ambient noise at these frequencies in certain times of the year.

Modelling studies of the likely response of the auditory structures in the inner ears of baleen whales based on their anatomy and dimensions of organs and the mechanical characteristics of their tissues provide further indication that baleen whales are most sensitive at the lower frequencies at which they vocalise (e.g. Ketten, 1997; Houser *et al.*, 2001). Neither this approach, nor the inferences based on the frequency range of the animals' vocalisations provide any indication of absolute sensitivity however.

A thorough review of the literature on baleen whale hearing and approaches to infer it, is provided in Christine Erbe's report for Defence Research and Development Canada (Erbe, 2002).

Table 4. Characteristic frequencies of vocalisations produced by baleen whales (Richardson *et al.* 1995; Matthews *et al.* 1999).

Species	Frequency range of tonal vocalisations (kHz) (mean minimum – mean maximum)
Minke whale	0.06 – 0.14
<i>Balaenoptera acutorostrata</i>	
Humpback whale	0.25-4
<i>Megaptera novaeangliae</i>	
Fin whale	0.015 – 0.043
<i>Balaneoptera physalus</i>	
Blue Whale	0.017 – 0.019
<i>Balaenoptera musculus</i>	

Hearing sensitivity of toothed whales

Behavioural audiograms have been reported for some smaller odontocete species mainly dolphins and porpoises (Table 5). These species are most sensitive to sounds above about 10 kHz, below which point sensitivity deteriorates. High frequency hearing is good; upper limits of sensitive hearing range from about 65 kHz to well above 100 kHz. This reflects the use by these species of high frequency sound pulses for echolocation and moderately high frequency calls for communication.

Frequencies at which the species in Table 5 had most sensitivity ranged from about 8 to 90 kHz and here their hearing is acute with the lowest underwater thresholds of any marine animals. Below the frequency range of optimum sensitivity, thresholds increase gradually with decreasing frequency.

Hearing sensitivity has not been measured in the majority of the larger odontocetes including sperm whales, pilot whales and most of the beaked whales. However, ABR methods were used to make an assessment of the frequency sensitivity of a stranded Gervais' beaked whale (*Mesoplodon europaeus*) by

(Cook et al., 2006). They found best sensitivity between 40 and 80kHz, a band which covers the emphasised frequencies in beaked whale vocalisations.

Table 5. Hearing sensitivity of toothed whales from underwater audiograms

Species	Lowest Frequency tested (kHz)	Threshold (dB re 1 μ Pa)	Most sensitive Frequency (kHz)	Threshold (dB re 1 μ Pa)	Upper Frequency (kHz)	Threshold (dB re 1 μ Pa)
Killer whale	1	105	20	34	100	75
Bottlenose dolphin	0.075	130	60	47	150	135
Risso's dolphin	1.6	124	8.0	63.7	110	123
Harbour porpoise	0.25	115	100	32	180	106

For those species occurring in the SEA 8 area for which data on hearing sensitivities are not available, the frequency range of assumed reasonably acute hearing (based on characteristic frequencies of vocalisations) is shown in Table 6.

Small odontocetes have lower hearing thresholds at higher frequencies than phocid seals. At their best frequencies, odontocetes are around 20-30 dB re 1 μ Pa more sensitive than are phocids. However, below about 2 kHz phocids become more sensitive than small odontocetes. For example, at 2kHz harbour porpoises and juvenile bottlenose dolphins have hearing thresholds of 50-70 dB re 1 μ Pa, similar to measures for a range of phocid seal species. At 100Hz, dolphin hearing thresholds had risen to 130 dB re 1 μ Pa. At 100Hz, harbour seal threshold was estimated to be 95dB re 1 μ Pa, approximately 35dB better than the dolphin. Many of the man-made sounds in the sea are within this low frequency band.

3.1.1.2 Zone of responsiveness

This is defined as the region around a source within which a marine mammal shows an observable response (Richardson *et al.* 1995). Behavioural responses are always inherently variable. While the physical process of detecting or being damaged by a sound can be predicted reasonably reliably from a combination of empirical studies and acoustic models, this is not the case for behavioural response. The reactions of an intelligent marine mammal to a particular stimulus may be affected by several factors, e.g. nutritional state (hungry or satiated), behavioural state (foraging, resting, migrating etc.), reproductive state (pregnant, lactating, juvenile, mature), location and of course by conditioning from previous exposure history.

Table 6. Characteristic frequencies of vocalisations produced by other toothed whales found in the SEA8 area (Tonal data taken from Matthews (1999) click data from Rasmussen (2002), Hooker (2002) and Johnson (2004). na = not applicable – tonal vocalisations not known; * data from Pacific white-sided dolphin *L obliquidens*); ** broad band with no clearly defined peak frequency

Species	Frequency range (mean minimum to mean maximum) for whistles (kHz)	Peak Frequency Clicks
Long-finned pilot whale <i>Globicephala macrorhynchus</i>	–3-6	No data
Sperm whale <i>Physeter macrocephalus</i>	na	10-20
Northern bottlenose whale <i>Hyperodon ampulatus</i>	na	24
Cuvier’s beaked whale <i>Ziphius cavirostris</i>	na	30-50
Bottlenose Dolphin <i>Tursiops truncatus</i>	5-16	52
Whitesided Dolphin <i>Lagenorhynchus acutus</i>	8-12	59*
White-beaked dolphin <i>Lagenorhynchus albirostris</i>	9-12	120
Common dolphin	6-12	**
Harbour Porpoise	na	120

To date there have been a number of observational studies which have documented changes in distribution and behaviour of marine mammals in the presence of acoustic stimuli. For practical and political reasons, these have usually involved studies of large cetacean species. Thus, in their comprehensive review of marine mammals and sound, Richardson *et al.* (1995) devoted 15 pages to the responses of cetaceans to ships and boats and only two pages to the reactions of pinnipeds.

One of the best known examples of noise inducing an acute and serious effect on marine mammals is the mortality of beaked whales as a result of mass stranding associated with the use of military sonar (see below). While the causal association between the use of mid-frequency sonar and these dramatic incidents is now accepted, the mechanisms that lead to these mortalities have yet to be established. Recent observations suggest that these animals may have developed decompression sickness (Jepson *et al.* 2003; Fernandez *et al.* 2005) and it has been proposed that this could be induced when the diving behaviour of animals is altered in response to sonar signals. For example, animals disturbed by sonar may surface too quickly and/or remain too long at the surface. While, in the absence of direct observations during exposure to sonar signals, this mechanism remains hypothetical, the example does serve to emphasise that behavioural changes in response to acoustic signals can have acute and serious consequences.

Available information on behavioural and physiological responses of seals and cetaceans, to some of the potential noise sources in the SEA 8 area are described below.

3.1.1.3 Zone of masking

To be audible, a sound must be detectable against the background noise. Thus, the level of background noise will often determine whether a sound is detectable or not, especially at frequencies where the animal's hearing is highly sensitive. Richardson *et al.* (1995) suggest that, as a rule of thumb, a mammal can barely detect a sound signal if its received spectrum level¹ is equal to the level of noise in the 1/3 octave band in which it lies.

Critical ratios, i.e. the ratio of sound level to background noise level at which detection is masked, have been estimated for a range of species. These have usually involved high frequency or continuous tone sound sources (Southall *et al.* 2000; Richardson *et al.* 1995). For harbour seals, Turnbull and Terhune (1993) showed that increasing repetition rate decreased hearing threshold for pulsed sounds above 2kHz irrespective of the level of masking, i.e. faster repetition decreased the critical ratio. This implies that critical ratios for irregular short pulses will be higher than for continuous tones. To date there are no direct data on the masking effects of background noise on ability to detect low frequency pulsed sounds.

The efficient detection of a wide range of sounds is biologically important for marine mammals. These will include sounds made by conspecifics, prey and predators, environmental noise useful for orientation and navigation, and, for echo-locating species, the echoes returning from ensonified objects. Masking by noise will decrease the maximum range at which these activities can take place. A useful way to think about the significance of masking for an animal is in terms of the reduction it causes in the efficiency with which these activities can be performed. Where a directional sound beam is produced, in the case of echolocation for example, the proportional decrease in effective range will be the most appropriate metric. For other acoustic tasks, for example detecting the calls of a conspecific, it is the decrease in the effective sensory area or volume that should be considered. Mohl (1981) modelled masking effects in such a framework. He found that proportional decrease in detection range was independent of the signal to noise ratio necessary for a particular task and that it was inversely related to the amount of background noise already in the environment. Even low levels of anthropogenic noise can significantly decrease the efficiency with which acoustic tasks can be performed, especially in regions that have low levels of "natural" background noise.

Masking effects have not been investigated in large cetaceans. However, as these species tend to produce lower frequency vocalisations we can assume that they will be most affected by low frequency noise.

Masking has not been studied directly in marine mammals in the field. However, there are examples of marine mammals changing their vocal behaviour in ways which would reduce the effects of masking. For example, increasing source levels, increasing frequency (to reduce overlap with low frequency noise) and increasing vocalisation rate to increase signal redundancy (e.g. Miller *et al.* 2000).

3.1.1.4 Zones of hearing loss and injury

In terrestrial mammals, exposure to loud sounds can lead to temporary threshold shifts (TTS), permanent threshold shifts (PTS) and even non-auditory tissue damage, which may be fatal. For continuous sound sources, the intensity of the signal relative to the hearing threshold at that frequency, and the duration of the exposure can both affect the timing of the onset of TTS and PTS. As a general rule, if a sound can cause TTS, a prolonged exposure to it will lead to PTS. For impulsive sounds, the intensity, rise time, pulse duration, pulse repetition rate and duration of exposure can all affect the timing and extent of TTS and PTS (Richardson *et al.* 1995). In the case of extremely loud sounds there may be an instant PTS and even damage to non-auditory organs.

Hearing loss

Only recently, have experiments to induce threshold shifts, been conducted on captive marine mammals. (Schlundt *et al.*, 2000) measured the levels of intense tones required to cause a 6dB reduction in masked hearing threshold in two beluga and five bottlenose dolphins. To provide a more or less constant noise floor in the uncontrolled study location, San Diego Bay, an environment with significant and variable ambient noise levels, masking noise was broadcast as a background during experiments. Hence "masked thresholds", not absolute thresholds were measured and it should be noted that shifts in masked thresholds are generally smaller than the non-masked TTS that would be induced by the same level of fatiguing

1. $\delta\text{B re } 1\mu\text{Pa}^2/\text{Hz}$.

noise. One second tones centred at 0.4, 3, 10, 20, and 75 kHz were used as fatiguing noises in this experiment. At 10 and 20kHz received levels of 192dB were required to cause a 6dB mTTS.

Au *et al.*, (1999) subjected individuals to a 5-10kHz, octave band, fatiguing source for at least 30 minutes over a one hour period to explore the effects on bottlenose dolphins of longer exposures to broader band noise. They found no TTS at a received level of 171dB but a threshold shift of 12-18dB occurred at 179dB re 1 μ Pa.

Lucke *et al.*, (2007) report on recent work measuring TTS in harbour porpoise. This is particularly relevant to this SEA both because the species they studied is, the commonest inshore cetacean in this area and because the fatiguing signals they used were low frequency pulses which are particularly relevant to activities such as seismic surveys and pile driving. Lucke et al exposed a porpoise to pulses from a small airgun as a surrogate for pile driving noise and found that measurable TTS was induced after exposure to a single airgun pulse with a received sound level of 184dB re 1 μ Pa p-p, and a received energy of 165dB re 1 μ Pa²s. These are much lower exposure levels than other researchers have reported and it is also unexpected to find a species which is a high frequency specialist so vulnerable to low frequency pulses.

TTS has been induced, experimentally, in three pinniped species, harbour seal, northern elephant seal and Californian sea lions (Kastak and Schusterman, 1996; Kastak *et al.* 1999). All three species showed a similar TTS of 4.6-4.9 dB, after 20-22 minutes of exposure at 65-70 dB above threshold level in the frequency range 0.1-2 kHz.

With the absence of reliable information on the levels of sound likely to cause hearing damage in most marine mammal species, it has been common practice to apply human Damage Risk Criteria (DRC) to other mammals (Richardson *et al.* 1995). Empirical studies have shown that humans exposed, in air, to continuous sound levels 80dB above their absolute hearing thresholds are likely to suffer TTS and eventual PTS. If this DRC is applied to marine mammals we would predict that at low frequencies (<500 Hz) TTS would occur at around 165-180 dB re 1 μ Pa in phocids and at around 180-210 dB re 1 μ Pa in small odontocetes.

These represent the DRC for exposure to continuous noise. For intermittent sounds, e.g. airgun blasts, the sound levels may be significantly higher, and will depend on the length and number of pulses received. Richardson *et al.* (1995) estimated the DRC for 100 pulses to be 138 dB above absolute hearing threshold. This would be approximately 208 dB for a harbour seal and would be higher for small odontocetes. Such levels could be encountered within 100m horizontally from a large commercial airgun array.

It must be stressed that the validity of applying DRC derived from human studies to seals and odontocetes is unproven, though the recent TTS studies mentioned above suggest that this is not an unduly conservative assumption. Given the lack of information on threshold levels for large cetaceans it is not possible to suggest reliable DRCs for this group.

One example of noise induced damage highlights the problem of our lack of knowledge. Mass strandings of Cuviers's beaked whales in other areas linked to the use of powerful sonars had suggested that this species, and perhaps beaked whales generally, are particularly vulnerable to being damaged by such sound sources (Frantzis *et al.* 1998). Whales killed in recent well documented, standing events in the Bahamas and the Canaries exhibited physical damage to a variety of structures associated with hearing and/or adjacent to air spaces and symptoms consistent with decompression sickness (Balcomb, 2001; Evans and England, 2001; Jepson *et al.* 2003; Fernandez *et al.* 2005). It now seems likely that military sonar has been causing beaked whales to strand regularly since the sixties. This phenomenon is a cause for more general concern for several reasons:

1. Our knowledge of the anatomy and vocal behaviour of beaked whales provide no indications of their apparent vulnerability to noise;
2. Other species may be equally vulnerable, and this group may be vulnerable to other intense noise sources;
3. The mechanism that led to the injury and damage in these animals still remains unknown.
4. Although, with hindsight mass strandings can be seen to have been linked in time and space with sonar deployments, it has taken 40 years for the association to be recognised and accepted.

Non-auditory effects

Blast injury

Very intense pressure waves, e.g. blast waves from explosions, have the potential to cause damage to body tissues. Damage is most likely to occur where substantial impedance differences occur, e.g. across air/tissue interfaces in the middle ear, sinuses, lungs and intestines.

Blast damage in marine mammals has been investigated using both submerged terrestrial mammals (Goertner, 1982; Richmond, Yelverton *et al.* 1973; Yelverton, Richmond *et al.* 1973) and dolphin cadavers (Myrick, Cassano *et al.* 1990). Goertner (1982) estimated the distance at which slight lung and intestinal injuries would occur in various marine mammals. Marine mammals are at greatest risk of injury when they are at the same depth as, or slightly above, the explosion. Risks drop off quite sharply above and below this depth. E.g. a harbour porpoise within 750m of an explosion of a 545kg charge at 38m is likely to suffer injury if it is at the same depth. But 30m above, or 43m below it, the range for injury is predicted to reduce to 500m. "Safe" distances for larger animals are expected to be shorter than for smaller ones (Richardson *et al.* 1995). Young (1991) estimated safe ranges for marine mammals of three different sizes and for human divers. However, the "safe" distances for humans are substantially greater than those for an equivalent sized marine mammal. Richardson *et al.* (1995) have suggested that a precautionary approach would involve applying the human value for all marine mammals. This would give a safe distance of 600m for a 1kg explosion, 900m for a 10kg explosion and 2km for a 100kg explosion.

Small explosive charges have been used to try to keep seals and small whales away from fishing gear, but with limited success. Humpback whales did not apparently move away from a construction site off the coast of Newfoundland where very large charges (200-2,000 kg) were used in construction work (Lien *et al.* 1993). However, two whales with severely damaged ears became entangled in fishing gear during this time, and it seems very likely that the explosions were at least partly responsible for their deaths (Ketten *et al.* 1995). Five of eleven Weddell seals sampled in the vicinity of blasting sites showed signs of inner ear damage (Bohne *et al.* 1985, 1986) and various otariid seals have been observed to be killed directly by explosives (Fitch and Young, 1948; Trasky, 1976). Thus it seems that serious damage can result even in cases where the behaviour of marine mammals is not dramatically affected, and they may remain in areas where damaging blasting is taking place.

It isn't clear whether intense sound sources, such as seismic airguns or military sonar, could cause tissue damage. If so, this would be at very short range and small numbers of animals would be affected severely.

Other effects

Air filled cavities within the body may be made to vibrate by intense, continuous wave underwater sound. Effects will be most marked at frequencies close to their resonant frequencies, which may vary with dive depth.

Human divers exposed to intense low frequency sound report feelings of vibration, discomfort and disorientation which may be linked with over stimulation of the vestibular system. It is likely that some of the effects reported by divers also occur in marine mammals. If so, they are likely to be evinced as behavioural disruption and disorientation.

Intense sound fields may also cause gas bubbles to develop around micronuclei within tissues. This could be a major concern for human divers whose body tissues become super-saturated from breathing compressed gasses during dives. Marine mammals do not breath compressed air, but the repetitive nature of their diving may lead to super-saturation (Ridgway and Howard, 1982; Houser, Howard and Ridgway, 2001).

Crum and Mao (1996) modelled the process of bubble growth in sound fields and concluded that a few minutes of exposure to 190 dB re 1 μ Pa in the frequency range of 250-1000 Hz, could induce bubble formation which might lead to occlusion of capillaries. Thus, exposure to intense sound could be the critical factor triggering the bends in human divers or marine mammals with super-saturated tissues.

The observation of symptoms consistent with decompression sickness in beaked whales that stranded during a sonar related incident in the Canaries has led to speculation that sound exposure may lead to

decompression sickness in cetaceans at lower received levels, perhaps by disrupting patterns of diving behaviour (Jepson et al. 2003).

3.1.3. Noise Exposure Damage Criteria

Managing offshore activities to minimise risk to marine mammals requires agreed criteria for levels of risk which are unacceptable and clear criteria or levels of acoustic exposure above which animals are at risk. No clear levels or criteria have been agreed in UK or Europe. However, a group of 13 experts in marine mammal and underwater acoustics from the USA and Canada, who worked together over several years to review the literature and derive a set of criteria for marine mammal noise exposure, have recently reported their findings (Southall *et al.*, 2007). To derive these noise exposure damage criteria they divided marine mammal species into five broad groups based on their functional hearing characteristics. These groups are 1. baleen whales, low frequency cetaceans; 2. Some of the larger odontocetes which are designated as mid-frequency cetaceans; 3. high frequency odontocetes, comprising all the small odontocetes and dolphins, 4. the pinnipeds and 5. pinnipeds listening in air. For each of these groups they derived broad acoustic filters (M_0 filters) reflecting information and inference about the hearing sensitivity of the members of the groups. These filters were then applied to any sound of interest for assessments of damage criteria based on sound exposure levels. Sound pressure level assessments were made using flat filters. The M_0 filters proposed here were broad and flat-bottomed and in effect merely defined the animals function hearing range; they weren't closely based on particular audiograms as some authors (e.g. Nedwell *et al.*, 2007) have proposed. The audiogram approach is better suited to assessing animal responses to barely audible sounds rather than vulnerability to damage from intense ones. Southall *et al.*, also considered several different noise types: single pulses, multiple pulse and continuous noise. For each noise type and each species group they developed damage criteria for both sound levels and for sound exposure levels, the total energy dose that could build up over evened or multiple exposures. Table 7 summarises some of the proposed criteria for unacceptable exposure.

Table 7: Proposed injury criteria proposed by Southall *et al.* (2007) for exposure of individual marine mammals to discrete noise events within a 24 hour period. Sound pressure criteria are based on peak pressure levels in flat, unfiltered signals. Sound exposure criteria are made after the filter deemed appropriate for the particular species group have been applied.

Marine Mammal Group	Criterion	Sound Type		
		Single Pulses	Multiple Pulses	Non Pulses (continuous)
Low Frequency Cetacean	Sound Pressure	230dB re 1 μ Pa (peak)(flat)	230 dB re 1 μ Pa (peak)(flat)	230 dB re 1 μ Pa (peak)(flat)
	Sound Exposure	198 dB re 1 μ Pa ² s (M_{lf})	198 dB re 1 μ Pa ² s (M_{lf})	198 dB re 1 μ Pa ² s (M_{lf})
Mid Frequency Cetaceans	Sound Pressure	230 dB re 1 μ Pa (peak)(flat)	230 dB re 1 μ Pa (peak)(flat)	230 dB re 1 μ Pa (peak)(flat)
	Sound Exposure	198 dB re 1 μ Pa ² s (M_{mf})	198 dB re 1 μ Pa ² s (M_{mf})	215 dB re 1 μ Pa ² s (M_{mf})
High Frequency Cetaceans	Sound Pressure	230 dB re 1 μ Pa (peak)(flat)	230 dB re 1 μ Pa (peak)(flat)	230 dB re 1 μ Pa (peak)(flat)
	Sound Exposure	198 dB re 1 μ Pa ² s (M_{hf})	198 dB re 1 μ Pa ² s (M_{hf})	215 dB re 1 μ Pa ² s (M_{hf})
Pinnipeds	Sound Pressure	218 dB re 1 μ Pa (peak)(flat)	218 dB re 1 μ Pa (peak)(flat)	218 dB re 1 μ Pa (peak)(flat)
	Sound Exposure	186 dB re 1 μ Pa ² s (M_{pw})	186 dB re 1 μ Pa ² s (M_{pw})	203 dB re 1 μ Pa ² s (M_{pw})

Some caveats should be noted. Some of the panel's work necessarily rests on some very uncertain data and the criteria they proposed have not been independently tested. It is interesting to note that neither the apparent sensitivity of beaked whales to military sonar nor Lucke *et al.* (2007)'s observations of TTS in porpoise after exposure to single low frequency pulses at relatively low levels, are predicted or explained by the proposed criteria. Even so, this paper is an enormously useful one. It provides an in-depth and expert review of the available literature but its greatest value lies in the fact that the procedures followed to derive the criteria are so clearly and transparently explained, making it possible for others to recalculate these if and when improved data become available and to calculate criteria which are appropriate to particular regulatory and legislative frameworks.

3.1.2 Behavioural responses of marine mammals to different types of noise

Many offshore activities are noisy. Two that are of particular concern offshore in the SEA 8 area are offshore oil and gas exploration and production and the construction and operation of wind farms. These activities involve a number of distinct phases which produce a range of loud and potentially disturbing and or even damaging sounds. Knowledge of noise production and marine mammal responses associated with offshore oil and gas is well documented, however, with the emergence of renewable technology comes potentially new impacts on the marine environment. Offshore wind farm development is still relatively new, however our knowledge of the potential impacts from activities associated with this industry are slowly improving (Thomsen *et al.* 2006; Madsen *et al.*, 2006b; Gordon *et al.*, 2007).

3.1.3.1 Oil and Gas

Three phases in the life of an oil and gas field can be identified

- **Exploration** (Seismic Survey, sidescan sonar),
- **Extraction** (Drilling, FPSO vessels, dynamically positioned vessels, sonar surveys, seismic site surveys, increased boat traffic, pipeline laying)
- **Decommissioning** (Explosive removals)

We very briefly describe some of the known and potential effects of noise and how these relate to various stages in the life of offshore oil and gas fields. We then try to identify the key knowledge gaps and prioritise the research needed to close them.

Seismic surveys

Exploration for oil and gas reserves usually requires a series of seismic surveys to characterise the sub-surface rock formations. This involves generating a series of high-energy acoustic pulses in the water column. Sound pressure waves penetrate the seabed to produce seismic waves. By measuring the strength and time of arrival of reflected signals geophysicists can map the patterns of the reflective boundaries between different rock strata.

Airgun arrays are currently the commonest high-energy source used for seismic survey; by 1985 more than 97% of marine seismic surveys used airguns (Turnpenny and Nedwell, 1994). Airguns produce sound pulses by rapidly venting high-pressure gas from a chamber. The resulting oscillating bubble produces a series of pressure waves with a waveform that can be described as a damped cosine, with a reduced amplitude and slight delay in the initial peak (Malme *et al.* 1986; Turnpenny and Nedwell, 1994; Barger and Hamblen, 1980). Airgun arrays are towed behind purpose-built survey vessels. Guns are suspended at depths of 1 to 10 m and fired at intervals of several seconds, depending upon the speed of the survey vessel and the depth of the water. In general the boats travel at 4-5 knots ($2-2.5 \text{ m.s}^{-1}$) and guns are fired at roughly 10 s intervals. The length of any firing sequence is dictated by the individual survey requirements, but it is not unusual for firing sequences to continue for many hours.

With the exception of explosives, airgun arrays are the most intense man made sound sources in the sea. The peak levels of sound pulses are much greater than the root-mean-square (RMS) levels from continuous sources such as ship noise or other industrial sources (Richardson *et al.* 1995). However, because the sound pulses are short relative to the inter-pulse intervals, the total energy transmitted to the water may be lower than from some continuous sources. Direct comparisons between different types of sources are therefore difficult. Their ability to cause hearing damage will of course also depend on the characteristics of the receiver (marine mammal ears) which in many cases are poorly known. Broadband source levels of 248-259 dB re $1\mu\text{Pa}$ @1m are typical of large arrays (Richardson *et al.* 1995).

Airgun arrays are designed so that signals from individual guns interact to maximise the downward transmission of the acoustic energy. Pressure fronts from different points in the array, which constructively interfere in the vertical plane, are unlikely to do so in the horizontal plane. So, effective source levels for horizontal transmission will generally be lower than for vertical transmission and will depend critically on the geometry of the array and the position of the receiver relative to it. A linear array of guns will generally have a much lower effective source level in the direction of its axis than to the side.

While these horizontal transmissions are lower than the levels directed vertically, they are still very loud in absolute terms and relative to background levels. Estimated source levels for a 28.7 litre array at 'end-fire' aspect were 217dB re 1 μ Pa@1m, and would be expected to be greater at the sides (Malme *et al.* 1983). Thus, significant amounts of acoustic energy may be transmitted horizontally through the water column (Richardson *et al.* 1995). Goold and Fish (1998) detected sound levels above background, at ranges up to 8km from a 37 litre array and detection ranges of 100s of miles are not uncommon.

A recent paper by Madsen and colleagues (2006a) provides an interesting perspective on this question. They analysed airgun pulses recorded on an acoustic recording telemetry device attached to eight different sperm whales. Useful recordings were made at range from 1.4 – 12.5km and at depths from the surface to 658m. These served to emphasise how variable the received spectral and temporal characteristics of airgun pulses were and that complex deep water propagation meant that over some ranges received levels could actually increase with increasing distance from the source.

Most of the energy in airgun blasts is below 200 Hz. Barger and Hamblen (1980) reported a bandwidth of 40Hz centred about 120 Hz. The peak spectral level (the SPL in 1Hz bands) occurred between 35 and 50 Hz, and decreased monotonically with increasing frequency; spectral level at 200Hz was 48dB down on the peak at 40Hz.

Source levels at higher frequencies are low relative to that at the peak frequency but are still loud in absolute terms and relative to background levels. Goold and Fish (1998) recorded 8 kHz sounds above background levels at a range of 8km from the source, even in a high noise environment.

The now extensive literature on the effects of seismic surveys on marine mammals has recently been reviewed by Gordon *et al.* (2004). See also recent reviews by Nowacek *et al.*, (2007) and Weilgart, (2007).

The reactions of some baleen whales (bowhead, grey, blue, fin, minke and humpback) to airgun noise have been studied in the field (summarised in Gordon *et al.* 2004, table 2). Clear behavioural responses, in terms of changes in surfacing patterns and movement away from the source when it was within 5 km of the whales, have been observed on a number of occasions (Malme *et al.* 1983, 1984, 1988; Richardson *et al.* 1995). Reactions have been most pronounced when the whales were to the side of the arrays long axis. McCauley *et al.* (1998) showed consistent avoidance of airguns by humpback whales during a series of careful observations made in Australia. They found that mothers and calves were more vulnerable to disturbance than single animals. Fin and blue whales continued to call in presence of airgun noise (McDonald *et al.* 1993). But McDonald also showed apparent avoidance by fin or blue whale. In UK waters, minke whales were sighted significantly further away from seismic vessels during periods of seismic array activity, suggesting active avoidance (Stone 1997, 1998).

The hearing ability of toothed whales is relatively poor at low frequencies; nevertheless there is sufficient high frequency energy in the output of airgun to make them audible at distances of >10km. In addition, seismic arrays carry a network of high frequency transponders for positioning. Goold (1996) presented evidence which he interpreted as showing large scale, long term changes in abundance and distribution of common dolphins during a survey and shorter term changes in behaviour between periods when guns were on and off within a survey block. In a later paper (Goold & Fish, 1998), seasonal changes in the distribution of dolphins in the same area at the same time were revealed that may explain some, or all, of the larger scale changes previously attributed to seismic surveys. This demonstrates the difficulty of interpreting observational studies made from platforms of opportunity.

Stone (1997, 1998, 2000, 2001) summarised reports from marine mammal observers (MMOSs) working on seismic vessels operating around the British Isles in which white-beaked and white-sided dolphins were seen less often during periods of seismic array activity. Conversely, more pilot whales were seen during periods of activity. This may indicate different avoidance strategies for deep diving animals like pilot whales. Sperm whales have been reported to stop calling and/or move away from distant airgun

noise (Mate *et al.* 1994; Bowles *et al.* 1994). However, other observations suggest that sperm whales show rather little response to airguns (Swift *et al.* 1999; Madsen *et al.* 2006).

Both harbour and grey seals showed short-term avoidance behaviour during controlled exposure experiments with small airguns (Thompson *et al.* 1998). In both cases seals abandoned foraging sites and swam away from airguns but returned to forage in the same areas on subsequent days. By contrast, Harris *et al.* (2001) making observations from a seismic vessel operating in a shallow lagoon system in the Canadian Arctic, found no significant change in sightings rate between firing and non firing periods. Mean radial distance to sightings did increase, suggesting some local avoidance behaviour.

4D or time lapse seismic is rapidly becoming an accepted tool for reservoir management (Bouska and Donovan, 2000; Parker, Bertelli and Dromgoole, 2003; Koster *et al.* 2000). Data from sequential seismic surveys are compared, and differences between these “time lapse” datasets can be interpreted in terms of changes in the reservoir due to extraction activity. In addition, smaller scale “site surveys” may be made throughout the life of some oil fields. The effects of such repeated surveys are not known, but minor or even insignificant transient effects may become important if disturbance is repeated and/or intensified.

Vessel noise

There is substantial medium sized commercial and military shipping activity in this area. Noise from shipping is roughly related to vessel size; larger ships have larger, slower rotating propellers, which produce louder, lower frequency sounds. Various models for predicting shipping noise on the basis of speed and hull length have been developed and are summarised and compared in a review by Heitmeyer *et al.* (2004). Broadband source levels of ships between 55 and 85m are around 170-180 dB re 1 $\mu\text{Pa}@1\text{m}$ (Richardson *et al.* 1995), with most energy below 1 kHz. Use of bow thrusters increases broadband sound levels, in one case by 11 dB and includes higher frequency tonal components up to 1 kHz (Richardson *et al.* 1995).

Richardson *et al.* (1995) reviewed the published literature on the response of marine mammals to vessel noise. Many toothed whales appear to be tolerant of vessel noise and are regularly observed in areas where there is heavy traffic. Sperm whales have been reported to react to vessels with powerful outboard engines at distances of up to 2 km. Humpback whales and right whales are also reported to avoid large vessels in some areas. Fin whales are reputed to ignore large vessels, but they respond to close (< 100 m) approaches by whale-watching vessels by spending less time at the surface and by making shorter dives. In general, whales show very little response to slow approaches by vessels, but they may swim rapidly away from vessels producing sound which changes in intensity or head directly towards them. There is little or no data on the response of seals to vessel noise out at sea. The fact that so many large whales are struck and killed by shipping, indeed this may be a major factor preventing the recovery of North Atlantic right whale populations, is testament to the fact that these animals don't always detect and respond appropriately to shipping (Laist *et al.* 2001; Nowachek *et al.* 2004). Increased shipping associated with offshore activities will increase the risk of ship-strike mortality for larger cetaceans.

Drilling noise

Drilling noise is generally low frequency, with highest levels being recorded from drill ships. Conventional drill platforms produce very low frequency noise, with strongest signals at around 5 Hz whereas drill ships are reported to produce noise with tonal elements up to 600 Hz (Richardson *et al.* 1995; Greene, 1987). However, many different processes are involved in drilling oil wells and the noise emissions associated with each of these have been poorly classified and characterised. There may also be substantial differences related to the water depth, and whether the drill platform is floating or jacked up above water.

There are few data on the reactions of marine mammals to drilling noise. Studies of grey and bowhead whales during migration suggest that they are generally tolerant of low level drilling noise from drill ships, but show some avoidance behaviour when sounds are loud (>20 dB above background) (Richardson *et al.* 1985, 1990; Wartzok *et al.* 1989). Bowhead whales apparently reacted more to playbacks of drilling noise than to real operational sounds. Migrating Grey whales have been shown to change course to avoid drilling noise (Malme *et al.* 1983, 1984).

There is no clear evidence of avoidance behaviour by small odontocetes to drilling noise. Bottlenose, Risso's and common dolphins were seen close to oil platforms in the North West Atlantic, and sightings rates were similar in areas with and without rigs (Sorensen *et al.* 1984).

There is no evidence that phocid seals avoid drilling platforms. Both bearded and ringed seals approached a simulated drilling sound source, coming within 50m of the source (Richardson *et al.* 1995).

Construction activities associated with establishing new platforms and pipelines will also generate noise. The loudest sounds are likely to be impulsive hammering sounds, associated with pile driving and pipe installation. Source levels can be high, levels of 131-135 dB re 1 μ Pa. were measured 1km from a hammer used for pipe installation on an artificial island (Richardson *et al.* 1995) and much higher levels have been reported recently during the construction of wind farms, see later. Such impulsive sounds have similar frequency components to those generated by airguns. There are no available data on effects of pile driving noise on marine mammals.

3.1.2.2 Wind farms

Somewhat similar phases can be identified in the operational life of a wind farm.

- **Site Survey** (Seismic Survey, sidescan sonar),
- **Construction** (vessel traffic, pile driving in many cases, dredging)
- **Operations** (Turbine noise)
- **Decommissioning** (Possible Explosive removals)

Nedwell and Howell (2004) review likely noise sources at windfarms during these different phases. Geophysical site survey work would probably involve boomers and sparkers. These are less powerful than the seismic arrays used during oil and gas exploration but there is little information on their source levels or other acoustic characteristics.

The construction phase will often involve pile driving of monopiles and dredging activity. Pulses produced during pile driving of the large pylons used to support wind turbines can be very intense. For example, during construction of the Burbo Bank offshore windfarm in 2006 (Parvin and Nedwell, 2006) measured impact piling noise from 4.7m diameter piles and computed a likely source level of approximately 249 dB re 1 μ Pa @ 1m. Larger diameter, 6.5m piles, are planned for future constructions and it has been calculated that source levels for them will be some 8.5dB higher (Parvin *et al.*, 2006).

During a piling operation pulses are produced at a rate of 30-60 pulses per minute and several hours of continuous piling might be required for each pile.

Dredging may be required around foundations and in conjunction with cable laying. Suction and hopper dredgers have shown levels of up to 177 dB re 1 μ Pa in the range 80-200Hz.

No work has been carried out to investigate the marine mammal responses to wind farm construction in UK waters. However, reactions of marine mammals were studied during the construction of two windfarms in Denmark: at Nysted and Hons Rev. (Carstensen *et al.*, 2006; Tougaard *et al.*, 2006a; Tougaard *et al.*, 2006b) Research on harbour porpoise utilised automated passive monitoring devices, PODS, to collect data on distributions and relative densities. At both sites acoustic detection rates decreased substantially and significantly after pile driving with effects being observed at 10 and 15kms respectively. (These were the maximum ranges at which data were collected so the upper range at which effects occur has not been established.) At Hons Rev both a long term decrease in acoustic detections over the construction period as a whole and short term reactions to specific piling events were evident.

A year after construction had ceased densities at Horns Rev were not significantly lower than pre-construction levels. While at Nysted although detection rate increased after construction ceased levels were still significantly lower than during preconstruction monitoring.

There is little information on the responses of seals to such construction activity although a small sample of satellite tracked harbour seals continued to transit across Horn's Reef during construction work (Tougaard *et al.* 2003).

Both harbour seals and harbour porpoises showed behavioural responses to playback of underwater noise from a simulated 2MW wind turbine (Koschinski *et al.* 2003). Porpoises did not approach as close and vocalised more when the source was on, although the behavioural responses were less dramatic than those seen in response to a net pinger using the same approach. Harbour seals also appeared to move away from the source, although the increase in median closest approach distance was small, 120m to 180m.

There have been no direct investigations of hearing damage to marine mammals during piling activities. However, recent work in which a small airgun (simulating low frequency pulses from pile driving) was used to induce TTS in a captive harbour porpoise suggests that this high frequency specialist is more susceptible to hearing damage than had previously been assumed (Lucke *et al.*, 2007, see above). Gordon *et al.* (2007) used simple models of animal movement and propagation loss to investigate the risk of hearing damage to small cetaceans and seals in the vicinity of pile driving activities. Animals were assumed to move away from pile driving sources. These models indicated that the ranges at which damage might occur was dependent on a number of factors. Propagation conditions, which are difficult to model in typical wind farm sites areas and are rarely measured, were highlighted as being particularly influential. In many likely scenarios criteria for acoustic damage could be exceeded for animals at ranges of several thousand of meters, posing problems for conventional mitigation procedures.

3.1.3 Research Requirements

It is clear from earlier sections that current understanding of the effects of noise on marine mammals and the risks that this may cause is in most cases rudimentary. In most scenarios the main uncertainty is in the form of the relationship between observable responses and population consequences. However, there are legitimate grounds for concern and appropriate application of the precautionary principle will be required. Application of the precautionary principle in a situation with great uncertainty results in a restrictive management regime. Reducing uncertainty through focused research should allow the development of management schemes, which achieve conservation objectives while producing controls within which industry can operate. An appropriate risk assessment framework developed by Harwood (1999) for cetacean by-catch reduction can be applied to the marine mammal noise issue (Tyack *et al.* 2004). Without pre-judging the outcome of individual risk analyses we can identify broad areas of research, which are feasible and likely to be valuable.

- **Dose Response.** Research, often in the form of controlled exposure experiments, is needed to address key uncertainties about marine mammal acoustics, sensitivities to and effects of sound. The practical and ethical issues involved in designing and conducting controlled exposure experiments have been widely discussed within the marine mammal scientific community. An in-depth analysis of these issues has recently been presented by Tyack *et al.* (2004).
- **Exposure Risk.** Targeted surveys together with telemetry based studies of movements and behaviour of selected species should be linked with oceanography and monitoring of other components of the ecosystem to identify important habitats and explore why they are important and improve our ability to predict marine mammal distributions at sea, year round.
- Assessing **medium or long term consequences** of particular activities will require long term monitoring of the status and distribution of populations of interest. To be most useful this should be in place before new activities develop, i.e. managers must be pro-active in establishing monitoring. There are currently no monitoring schemes for any offshore cetacean populations in UK waters that would be capable of detecting even large changes in population levels. Achieving this cost effectively will require the development of new methods; passive acoustic techniques are one promising possibility for some species. Even with such programs, establishing direct cause and effect will be difficult and necessarily retrospective.
- **Development of effective mitigation.** Current mitigation practices are largely based on “common sense” measures and little work has been done to establish whether they work and/or could be made more effective. It will always be prudent to utilise effective mitigation measures, if they are easy to apply, even when harmful effects of noise have not been proven.

Addressing these knowledge gaps will require a substantial research program. Partnerships amongst noise producers (e.g. industry, renewables, shipping, military) should be established. While this may seem a daunting scientific task, it is, in reality, trivial compared to the engineering challenges that offshore engineers face and overcome every day.

3.2 Contaminants

3.2.1 Background

Marine mammals are exposed to a variety of anthropogenic contaminants. The main route for exposure is through their prey and as these mammals are top predators they are at particular risk from contaminants which biomagnify through the food chain (i.e. are found at increasing concentrations at higher trophic levels). Most research has focussed on two main groups of contaminants: the persistent organic pollutants (POPs) and the heavy metals. However, there is some information on other contaminants including the polyaromatic hydrocarbons (PAHs), the butyl tins and most recently the perfluorinated chemicals.

3.2.1.1 Persistent organic pollutants

This group of chemicals includes the organohalogenated compounds (such as the polychlorinated biphenyls - PCBs), the dichlorodiphenyltrichloroethanes (DDTs), polybrominated biphenyls (PBBs), polybrominated diphenyl ethers (PBDEs), chlordane, toxaphene, the cyclodienes (such as aldrin and dieldrin), and polychlorinated terphenyls (PCTs). Of these the occurrence and potential effects of the organochlorine compounds (OCs) are by far the best investigated. Many chlorinated pesticides are also included in this group. The significance of these compounds for marine mammals is that:

- they are highly lipophilic and hydrophobic.
- they differentially accumulate in the lipids of animals and are therefore sometimes found at high concentrations in marine mammal blubber.
- they are chemically very stable and persistent, many compounds being resistant to metabolic degradation.
- they are present as many different isomers and congeners, and comprise hundreds of different chemical formulations which may have different behaviours and toxicities.
- they have reproductive and immunosuppressive effects, and many are 'endocrine disrupters' - acting as hormone agonists or antagonists.
- animals are exposed to complex mixtures of compounds that may have additive or synergistic effects on various target organs and systems.

In marine mammals most of these compounds are sequestered into the blubber so much of the determination of POP residues has concentrated on this tissue. Between 90 and 95% of the total burden of many POPs, particularly PCBs and DDTs, are found in the blubber because of its high lipid content (Aguilar, 1985). The compounds are essentially bound away in this tissue until the lipid store is mobilised for energy requirements or for the production of milk. This aspect of the life cycle of marine mammals means they may be re-exposed to the contaminants when they call upon their blubber reserves during periods of natural fasting. This is particularly the case for animals that do not feed during the breeding season, and also means that females can offload a large proportion of their contaminant burdens to their offspring (Debiec *et al.* 2003). Other POPs may behave slightly differently and recent studies have shown the PBDEs to be at high concentrations in the adrenal glands as well as the fat stores (Klasson Wehler *et al.* 2001). These compounds, particularly the tetra and penta group, are now found in the blubber of seals and cetaceans from UK waters (Allchin *et al.* 1999) and in studies on juvenile grey seals, harbour seals, large and ribbon seals are associated with thyroid hormone disruption (Hall, *et al.* 2003; Chiba *et al.* 2001; Tabuchi *et al.* 2006).

Many factors can affect the occurrence and distribution of POPs in marine mammals. These include diet, foraging strategy, age, species, sex, nutritional condition. These confounding variables need to be considered when interpreting the significance of reported tissue concentrations (Aguilar *et al.* 1999). The large majority of persistent organic pollutants do not arise from oil exploration and production. However, there is currently concern over the impact of the polybrominated compounds (largely PBDEs which are used as flame retardants). The deca-product mixture is still in use, whilst the penta and octa- mixtures containing the lower brominated compounds (the congeners that have been found in birds, seabirds and marine mammals, De Wit, 2002) have been banned in Europe. In the US the penta and deca-mixtures are both still legally used in many industries but some States have now passed laws to phase out use of the penta and octa BDEs.

3.2.1.2 Heavy metals

The heavy metals are a heterogeneous group of compounds. Some are bioaccumulative (such as mercury) whereas others appear not to be (such as cadmium, chromium, nickel and copper). Data on zinc and lead in various species in the marine food web are equivocal (Muir *et al.* 1992). The liver, kidney and bone are the main target organs for heavy metals and levels can vary widely depending on the geographical location of the species. Marine mammals appear to be protected against the effect of many heavy metals because of the presence of metallothioneins (Bowles, 1999). These are proteins whose production is induced by the occurrence of divalent cations such as Hg⁺⁺, Cd⁺⁺, Cu⁺⁺ and Zn⁺⁺. Metallothioneins have a high affinity for binding such cations, and they sequester the metals to form biochemical complexities with reduced toxicities. In addition mercury forms complexes with selenium, producing insoluble tiemannite granules (Nigro *et al.* 2002). This is an important mechanism, complementary to excretion, and enables many species to cope with a relatively high dietary exposure to mercury (Dietz *et al.* 1996). High levels of liver cadmium have been reported in a number of cetacean species and this probably also reflects dietary preferences. High concentrations of cadmium are accumulated in the liver and gonads of cephalopods (Hamanaka *et al.* 1982) and Antarctic krill (Honda *et al.* 1987), the prey species of many cetaceans.

3.2.1.3 Polyaromatic hydrocarbons (PAHs)

The potential for the biomagnification of PAHs is low, because fish (the main food of marine mammals) are good metabolisers of PAHs compared with molluscs and other invertebrates. Bioaccumulation or exposure to these compounds will be lower in fish-eating marine mammals than those that feed on cephalopods or small crustaceans and plankton (such as the mysticete whales). Seals and cetaceans also have a detoxification enzyme system in the liver, which is induced in response to various xenobiotic compounds, including PAHs. This system (known as the mixed function oxidase, MFO or cytochrome P450 system) can convert parent compounds into excretable metabolites, largely by the addition of a hydroxyl group (Sipes and Gandolfi, 1991). This biotransformation of compounds may, however, be toxic if the metabolites produced are bioactive. In addition the rate at which transformation occurs is critical. If the non-toxic pathway is saturated, minor pathways, which produce further toxic intermediates, become involved. One isoform of the cytochrome P450 enzyme system is also called aryl hydrocarbon hydroxylase because it plays a role in the metabolism of PAHs. The regulation of certain cytochrome P450 enzymes involves a ligand-activated transcription factor known as the Ah (aromatic hydrocarbon) receptor (Timbrell, 1991). This has been investigated in a limited number of marine mammals but induction and activity of the cytochrome enzymes is widely used as a marker of exposure to inducers such as PAHs (Troisi and Mason, 1997; Mattson *et al.* 1998; Wolkers *et al.* 1999; Miller *et al.* 2005; Tilley *et al.* 2002) although their utility as a marker of exposure to OCs is questionable (Niimi *et al.* 2005).

3.2.1.4 Butyl Tins (Tributyl tin (TBT), Dibutyl tin (DBT) and Monobutyl tin (MBT))

These groups of compounds were identified in liver samples of marine mammals, following knowledge about their toxicity and endocrine disrupting effects in invertebrates and fish (Iwata *et al.* 1994). Results of analysis in liver samples from stranded animals have indicated a widespread contamination around the coasts of England and Wales; indeed TBT and DBT have been found in open ocean cetacean species, which indicates a wider contamination of the sea by these compounds (Law *et al.* 1999). However, recent data on temporal trends of DBT, TBT and MBT in harbour porpoises from Norwegian waters (Berge *et al.* 2004) have found a decrease in tissue concentrations following the restrictions on the use of TBT on small boats in the late 1980s. Nakata *et al.* (2002) found that TBT and its metabolites caused suppression of immune function (as measured by the proliferation of T lymphocytes) in blood samples collected from Dall's porpoises, bottlenose dolphins, a California sea lion, a spotted seal and humans at levels of around 90 ng/ml for TBT and DBT. When cells were exposed to a mixture of TBTs and PCB congeners the proliferative responses were suppressed even further, suggesting possible synergistic effects between these compounds.

3.2.1.5 Perfluorinated organochemicals

Perfluorinated organic compounds are widely used in the manufacture of plastics, electronics, textile and construction material in the garment, leather and upholstery industries. Recent studies have also found perfluorinated organochemicals (FOCs) in the tissues of marine mammals. Van de Vijver *et al.* (2003) measured the presence of FOCs in marine mammals, indicating a potential biomagnification of these

compounds and their widespread occurrence. Liver, kidney and spleen appear to be the major target organs (Van de Vijver *et al.* 2005). Among all the measured FOC compounds, PFOS (perfluorooctane sulfonate) was predominant in terms of concentration. The highest PFOS concentrations were found in the liver of harbour seal compared to white-beaked dolphin, harbour porpoise, grey seal, sperm whale, white-sided dolphin, striped dolphin, fin whale, and hooded seal. Harbour and grey seals and white-beaked dolphin, which displayed the highest trophic position, contained the highest PFOS levels, while offshore feeders such as sperm whales, fin whales, striped dolphin, and white-sided dolphin showed lower PFOS concentrations (Van de Vijver *et al.* 2005). A recent study on southern sea otters found high levels of perfluorooctanoic acid (PFOA) in their livers and concentrations of PFOA and PFOS were higher in animals that died of infectious disease than other causes (Kannan *et al.* 2006).

3.2.2 Sources of Data

There is a huge body of literature on contaminants in marine mammals worldwide. For example, the US Marine Mammal Commission (Long, 2000) issued a bibliography containing over 1,200 references and many more have been published in the last 6 years. However, there are many good reviews on the levels of contaminants found, the patterns of different compound groups in various species and the temporal changes in concentrations. The most comprehensive are: Aguilar and Borrell (1997), Geraci and St. Aubin (1990), Hall (2001), Law (1996), O'Shea (1999), Reijnders, Aguilar and Donovan (1999).

3.2.3 Knowledge

Although our knowledge of the effects of contaminants on marine mammals remains limited, largely due to the difficulties involved in investigating the responses in wild animals, it has increased considerably in recent years. It has been relatively straightforward to determine the tissue concentrations of various compounds in dead and live-captured animals, but the significance of these concentrations for the health and ultimate survival of the individuals has been more difficult to assess. Some studies have investigated the responses to exposure on animals in captivity, comparing responses between exposed and control groups and associations between dysfunction and contaminant exposure have been reported in free-living individuals and populations. These studies are increasing whereas those merely reporting levels in tissues are declining. Thus the body of information on correlations among toxic endpoints and contaminant exposure measures continues to increase and is now being supplemented with data from *in vitro* studies using cellular and molecular methods (De Guise *et al.* 1998; Hammond, *et al.* 2005a; Levin *et al.* 2005; Mori *et al.* 2006). In addition more recent work has also focussed on assessing the risk of contaminant exposure at the population level (Hall *et al.* 2006a; Hall *et al.* 2006b).

3.2.3.1 Persistent organic pollutants

Two observations on wild populations in the 1980s suggested that the uptake of POPs by marine mammals could have toxic effects similar to those reported in laboratory species. The first was the report that a serious decline in the population of harbour seals in the Wadden Sea might be due to the reproductive effects of contaminant exposure (Reijnders, 1980; Reijnders, 1984). Reijnders (1986) addressed this more directly in an experiment using captive harbour seals. Two groups of females were fed fish from different areas, one contaminated with OCs, the other much cleaner. Reproductive success was significantly lower in the group fed contaminated fish and failure was thought to occur at the implantation stage of pregnancy. The second effect was investigated following the outbreak of phocine distemper among harbour seals in European waters, in which differential mortality rates were reported among harbour seal populations around the UK coast (Hall *et al.* 1992a). This observation led to a study of the OC contaminant burdens among animals that were victims and survivors of the epidemic. The results suggested that animals that died of the disease had higher blubber levels of OCs than survivors, although it was not possible to control for all potential confounders (Hall *et al.* 1992b). This finding was also repeated in a study of contaminant burdens in striped dolphins following a similar outbreak of dolphin morbillivirus in the Mediterranean Sea in 1990 (Aguilar and Borrell, 1994) and in the 1987-88 bottlenose dolphin morbillivirus outbreak in the US (Kuehl *et al.* 1991). Furthermore similar results were obtained in live and dead harbour seals following the 2002 European PDV epidemic (Hall and Thomas, 2005). Studies by Ross *et al.* (1995) and DeSwart *et al.* (1994) found evidence for the mechanism of the effect. They reported immunosuppression in a group of captive harbour seals fed contaminated fish compared with animals fed clean fish. Natural killer cell activity (white blood cells that are particularly required in the defence against viral infection) in particular was depressed and lymphocyte function measured *in vitro* was lower in the exposed group. More recently Hammond *et al.* (2005a) found

that harbour seal immune function assays carried out *in vitro* were impaired when exposed to a commercial mixture of PCBs whereas grey seal (*Halichoerus grypus*) immunity was not affected.

The PBDEs (flame retardants) are being reported as potential endocrine disrupting compounds. Although the production and use of the lower brominated compounds has been controlled in Europe, the oil industry continues to use BDE209 and the penta-mixtures (commercial formulations with lower brominated compounds) are still used in North America. Hall *et al.* (2003) found a correlation between PBDEs and thyroid hormone levels in grey seals during their first year of life and in adult harbour seals (Hall and Thomas, 2007) but it is still unclear whether this relationship is causal. One particular flame retardant, hexabromocyclododecane (HBCD) is causing some concern and was found at relative high levels in the blubber of harbour porpoise stranded along the Irish sea coast, where levels were an order of magnitude higher ($\sim 3 \mu\text{g g}^{-1}$ lipid) than elsewhere except the northwest coast of Scotland where levels were $\sim 5 \mu\text{g g}^{-1}$ lipid (Zegers *et al.* 2006). But most importantly these compounds have been increasing in concentration in harbour porpoise stranded and by-caught throughout the UK (Law *et al.* 2006). Levels have more or less doubled in the last 10 years probably due to the use of HBCD as a flame retardant replacement for the PBDE mixtures whose use is now banned.

Bergman and Olsson (1985) also reported the occurrence of adrenocortical hyperplasia, hyperkeratosis and other lesions in grey and ringed (*Phoca hispida*) seals from the Baltic. The pathologies seen were indicative of a disease complex involving OCs and hormone disruption, a finding also demonstrated in laboratory animals (Fuller and Hobson, 1986). Other abnormalities associated with the highest exposures to PCBs include skull and bone lesions in grey seals (Bergman *et al.* 1992; Zakharov and Yablokov, 1990) and harbour seals from the Baltic (Mortensen *et al.* 1992).

An EU funded study known as BIO CET (<http://www.abdn.ac.uk/biocet/>) investigated the potential impact of POPs on reproduction in small cetaceans, pooling data from harbour porpoise and common dolphins found stranded in many countries around Europe. This included animals from the coast of Northern France (western English Channel between Normandy and Brest), probably representing animals using the Celtic sea (Pierce *et al.* 2008). They found that common dolphins from the French coast had the highest levels of total PCBs in their blubber and that $\sim 50\%$ of the sample ($n=36$) had levels above what is considered a potential critical threshold for reproductive effects ($17 \mu\text{g g}^{-1}$ lipid Kannan *et al.* 2000; based on the previous experimental data summarized above). Interestingly, animals stranding around the Irish coast, including those from southern Ireland, had the lowest levels, approximately 70% lower than the French animals. In terms of reproductive effects in these species, they found a high number of corpora albicantia (CA) in the ovaries, indicating infertility or a high number of miscarriages. Animals with relatively high blubber concentration of PCBs and PBDEs had the highest number of CA, which may be a cause or may be a consequence of this infertility (since animals are not able to offload some of their contaminant burden to their calf, their blubber concentration will remain high). For the harbour porpoise, the French animals again had the highest OC levels in their blubber but this was only based on two individuals in the sample. Indeed, animals from Scotland had higher blubber concentrations than those from the other locations with a sufficient sample size. In contrast to the common dolphins, in this species higher PCB concentrations tended to be associated with a lower number of CA in the ovaries.

Studies by Jepson *et al.* (1999, 2005) and Hall *et al.* (2006a) found that the risk of mortality from infectious disease in harbour porpoises that stranded around the coast of England and Wales increased with high exposure to PCBs (50% increase in relative risk at concentrations of total PCBs $>25 \mu\text{g g}^{-1}$ lipid in the blubber). In addition, stranded harbour porpoises from the German, North and Baltic seas were more severely diseased than by-caught animals and thymic atrophy and splenic depletion were significantly correlated to increased PCB and PBDE levels (Beineke *et al.* 2005). Various immune function endpoints measured *in vitro* in cetaceans (bottlenose dolphins Lahvis *et al.* 1995; beluga whales De Guise *et al.* 1998) and in wild polar bears (Lie *et al.* 2005) following PCB exposure further suggest that these compounds are also immunosuppressive to small cetaceans and bears.

3.2.3.2 Heavy metals

Of the toxic elements studied those of most importance are cadmium, lead, zinc and mercury.

Cadmium can sometimes be found at high concentrations in the livers of marine mammals (Law *et al.* 1991), but there does not appear to be any published information on cadmium-induced pathology in marine mammals. These high levels are probably due to naturally high cadmium concentrations in prey

species such as squid (Bustamante *et al.* 1998). Metallothionein sequestration appears to protect marine mammals from cadmium toxicity.

Lead is also found in many marine mammal tissues, particularly liver and kidney, but not at concentrations that are cause for concern (Law *et al.* 1991). Bone is a long-term storage target organ for lead, although again no associated histopathological lesions in have been reported. Smith *et al.* (1990) used isotopic ratios to show that the source of lead in some marine mammal species has shifted from naturally derived lead to anthropogenic aerosol-dominated forms.

Mercury can bioaccumulate through the food chain and is a well-recognised neurotoxin. Its interaction with selenium appears to be protective and various laboratory studies have shown that toxic effects of mercury were prevented or reduced by simultaneous exposure to selenium (Cuvin-Aralar and Furness, 1991). Some of the concentrations of mercury in the liver of marine mammals have exceeded those known to be toxic to other mammals but lethal effects have not been observed (Britt and Howard, 1983). Marine mammals seem able to metabolise mercury from its toxic methyl form found in fish. Although marine mammals can tolerate high concentrations of mercury immobilised as the selenide, methylmercury poisoning has been reported in a ringed seal an area of heavy industrialisation (Helminen *et al.* 1968). The recent study by Pierce *et al.* (2008) found the highest levels of mercury in the liver samples from common dolphins stranded along the French coast but these were not at concentrations high enough to cause concern.

Copper is an essential dietary element for mammals and a wide range of concentrations has been reported in marine mammals. In the UK levels of between 3 and 30 mg/kg have been measured in the liver of stranded animals and it has been suggested that this may represent the normal range of homeostatic control in marine mammals (Law, 1996).

Pillet *et al.* (2000) found that zinc exposure affected the phagocytic response of seal white cells *in vitro* and that this response differed between the sexes and Kakuschke *et al.* (2005) reported that a small number of harbour seals appeared to be hypersensitised to a number of heavy metals. Whilst there are few studies that show major impacts of heavy metals, it's possible that they may have combined effects as they often co-occur with the persistent organic contaminants.

3.2.3.3 Polyaromatic hydrocarbons (PAHs)

Polyaromatic hydrocarbons have rarely been studied in the tissues of marine mammals but where measurements in muscle tissue, liver and blubber have all generally been below 1µg/g. Law and Whinnett (1992) investigated PAHs in the muscle tissue of harbour porpoises stranded around the UK coast and found total PAH concentrations ranging from 0.11-0.56 µg/g wet weight and 0.47-2.4 µg/g wet weight Ekofisk crude oil equivalents. Specific PAHs were 2-4 ring compounds (naphthalenes, phenanthrenes, anthracene, fluoranthene and pyrene). Bond (1993) found similar compounds in the blubber of seals from the Moray Firth. The PAH levels in this species displayed large variations, with grey seals having higher levels than harbour seals (mean 15.78 (SD 25.54) µg/g dry weight in grey seals 2.67 (SD 5.77) in harbour seals).

The effects of PAHs on marine mammals are reviewed in Geraci and St Aubin (1990) and various responses from effects on the central nervous system, eyes and mucous membranes, thermal regulatory effects from fouling of fur, to induction of metabolic enzyme systems and effects on hormone levels were reported. These effects are largely observed following short-term acute exposure. Less is known about the effects of long-term chronic exposure. Although studies have shown that fish readily convert aromatic hydrocarbons to metabolites such as dihydrodiols and phenols (Krahn *et al.* 1984) and therefore fish-eating mammals may receive lower doses of parent PAHs, cetaceans which feed lower down the food chain are likely to be most at risk. In addition Neale *et al.* (2002) assessed the effects of the prototypic polycyclic aromatic hydrocarbon (PAH), benzo[a]pyrene (B[a]P), and two polychlorinated biphenyls (PCBs), CB-156 and CB-80, on the T-cell proliferative response to mitogen in harbor seal peripheral lymphocytes. They found a suppressive effect of B[a]P (10 uM) exposure on T cell mitogenesis. Exposures to 10 uM CB-156 and CB-80, and 1.0 and 0.1 uM B[a]P, did not produce significant depression in lymphocyte proliferation. Exposure to the model PAH at 10 uM resulted in a 61% (range 34-97%) average reduction in lymphocyte proliferation and they hypothesize that extensive exposure of PAHs by some marine mammals affects their cell-mediated immunity against viral pathogens.

The carcinogenic nature of certain PAHs, such as benzo(a)pyrene has been a concern. For example, Beland *et al.* (1993) reported the detection of benzo(a)pyrene adducts in DNA from Beluga whales in the Gulf of St Lawrence, but there is little evidence for the substantial exposure of marine mammals in UK waters to this compound. One of 27 UK harbour porpoises examined by (Law and Whinnett, 1992) between 1988 and 1991 was considered to have died as a result of a tumour.

Butyl tin compounds, largely tri- and di-butyl tin have now been reported in the liver and blubber of pelagic cetaceans and marine mammals in UK waters (Law *et al.* 1999), but no reports on their effects have been published.

3.2.3.4 Oil spills

Direct mortality from contaminant exposure has rarely been reported, and has usually been associated with major oil spills such as the *Exxon Valdez* in Alaska in 1989. High concentrations of phenanthrene (PHN) and naphthalene (NPH) were reported in the bile of oiled harbour seals (*Phoca vitulina*) collected following the spill (up to 23 times higher than in control seals) and high concentrations of PAHs in the blubber (up to 400 ppb) (Frost and Lowry, 1993). Due to the condition of many of the carcasses examined it was difficult to attribute cause of death to oil toxicity, but many animals exposed to oil did develop pathological conditions including brain lesions. Additional pup mortality was also reported in areas of heavy oil contamination when compared to unoiled areas.

More generally, marine mammals rely on their blubber for insulation and are thus less vulnerable than seabirds to fouling by oil (Geraci and St Aubin, 1990). However, they are at risk from hydrocarbons and other chemicals that may evaporate from the surface of an oil slick at sea within the first few days. Seals often barely raise their nostrils above the surface of the water when they breathe, so any seal surfacing in a fresh slick is likely to inhale vapours. Cetaceans also typically inhale close to the surface. Symptoms from acute exposure to volatile hydrocarbons include irritation to the eyes and lungs, lethargy, poor coordination and difficulty with breathing. Individuals may then drown as a result of these symptoms.

Grey and harbour seals come ashore regularly throughout the year between foraging trips and additionally spend significantly more time ashore during the moulting period (February-April in grey seals; August in harbour seals) and particularly the pupping season (October-December in grey seals; June-July in harbour seals). Animals most at risk from oil coming ashore on seal haul-out sites and breeding colonies are neonatal pups. These animals are born without any blubber and rely on their prenatal fur (the white lanugo in grey seals) and metabolic activity for thermal balance. They are therefore more susceptible than adults to external oil contamination (Ekker *et al.* 1992). Grey seals pups remain on the breeding colonies until they are weaned and unlike adults or juveniles, would be unable to leave the contaminated area. Females may also abandon contaminated pups during an oil spill, leading to starvation and premature death.

3.2.3.5 Oil dispersants

There have been no specific studies on the direct acute or chronic toxicity of oil dispersants to seals and cetaceans. The toxicity of oil spill dispersants to aquatic organisms under laboratory conditions appears to relate primarily to the chemical composition of the individual dispersant: for example, the type of solvent; its aromatic content (i.e. oil-based dispersants); the functional group(s) and molecular structure of the surfactants; their chemical stability; and concentration. Other factors that are important in oil spill dispersant aquatic toxicity are the duration of exposure of the organism, water temperature of the sea, oxygen content of the seawater, organism species/type, organism age, organism stage of growth/development, organism health. Indirect effects may occur if the prey items of marine mammals further down the food chain are affected.

3.2.4 Gaps in knowledge

With respect to the impact of oil exploration activities on contaminant exposure in marine mammals, no recent studies on the uptake of PAHs by marine mammals around the UK or pelagic cetaceans exist, and there is no information on the potential effects of long-term chronic exposure. Further studies are needed to determine current and background exposure levels in a variety of species and their prey, particularly prior to oil exploration and production activities within marine mammal foraging areas. In addition we still have no information on alkylated phenols in marine mammals. PAH sources from exploration and production are not now very significant (100 t/yr, OSPAR 2000) and most North Sea PAHs come from terrestrial combustion sources (> 7000 t/yr).

Information on the uptake and effect of polybrominated diphenyl ethers (the brominated flame retardants) on marine mammals is accruing, for a variety of invertebrates and fish as well as marine mammals, since higher levels were found in the UK than elsewhere in Europe (Zegers *et al.* 2001). Congener BDE209 is still used by the oil industry in the deca-mixture (containing 10 bromine atoms); this was found to be accumulated by grey seal pups from their prey in an experimental study (Thomas *et al.* 2005). However, this congener has not been found in marine biota to any great degree. However, there is concern that this fully brominated compound (containing 10 bromine atoms) can be degraded to form lower brominated compounds that are potentially toxic to marine mammals. Further research into the nature of the relationship between PBDE levels and thyroid hormones in seals is needed, including the full complement of thyroid active hormones. Some studies have reported a negative correlation in pups (e.g. Tabuchi *et al.* 2006) whereas others (Hall and Thomas, 2007) have found a positive relationship in adults.

Few investigations on contaminants in marine mammals have been able to address the effects at the population level. This is particularly important where, from dose-response studies, contaminants or mixtures of contaminants are likely to have effects on survival or fecundity. In particular we need to develop a framework in which the *population* risks can be evaluated. This has been investigated to some extent (Harwood *et al.* 1999) but more detailed empirical information is required. Most recently Hall *et al.* (2006b) developed an individual based model framework, using the impact of PCBs on bottlenose dolphins as an example of how to assess the effect of such compounds on population dynamics. This study and that of Schwacke *et al.* (2002) illustrate the need for reliable dose-response data for these and other species of marine mammal.

3.3 Disease

3.3.1 Background

It has long been known that marine mammals harbour large numbers of macroparasites, such as nematodes and cestodes as well as various ectoparasites (Margolis, 1954; Reijnders *et al.* 1982; Baker and Martin, 1992). However, these parasites usually do not cause severe harm unless the animals have an underlying primary disease or are stressed for other reasons.

There have been outbreaks of viral and bacterial disease epidemics among seals and cetaceans worldwide and these seem to have increased in frequency, particularly in the US, in recent years (Harvell *et al.* 1999). In UK and European waters major epidemics from phocine distemper (PDV) occurred in harbour and grey seals (PDV) in 1988 and again in 2002 (Hall *et al.* 2006c) and morbillivirus (DMV) occurred in Mediterranean striped dolphins in 1990 and US bottlenose dolphins in 1987 (Dietz *et al.* 1989; Jensen *et al.* 2002; Aguilar and Raga 1993; Lipscomb *et al.* 1994). This led to a number of studies into the epidemiology of morbilliviruses; for example investigations into the grey seals which is not susceptible to the disease as potential immune carriers that could account for the spread of the virus (Hammond *et al.* 2005b). These outbreaks were followed by other mass mortalities in the late 1990s, such as among Mediterranean monk seals, whose cause was disputed and although some evidence pointed to PDV as a cause (Osterhaus *et al.* 1997; Harwood, 1998; Hernandez *et al.* 1998) it seems more likely that this outbreak was due to algal toxin exposure.

Apart from such high profile, large-scale epidemic diseases, marine mammals are also known to suffer from a range of viral and bacterial infectious diseases.

3.3.2 Sources of data

A number of reviews of infectious diseases in marine mammals have been published and the major sources are given below: Dierauf and Gulland (2001); Van Bresseem, Van Waerebeek and Raga (1999); Harwood and Hall (1990); Visser, Teppema and Osterhaus (1991). Gulland and Hall (2007) recently reviewed the literature on diseases in marine mammals detailing how they have been investigated over time. This work resulted in a database of over 600 references which is available at the Sea Mammal Research Unit website (<http://www.smru.st-and.ac.uk>).

3.3.3 Knowledge

3.3.3.1 Viruses

Table 8 indicates the viral infections that have been reported among marine mammals. The morbilliviruses and influenza viruses have accounted for large scale mortalities around the world.

3.3.3.2 Bacteria

A range of organisms has been cultured from healthy and sick marine mammals and many are secondary infections in malnourished and starveling animals, particularly juveniles. Baker (1984) found that 40% of all grey seal pups died of infections such as peritonitis and septicaemia. *Corynebacterium* and *Streptococcus* accounted for the majority of infections and during the 1988 PDV epidemic *Bordetella* organisms were isolated from a large proportion of the sick animals but was not found in healthy individuals (Munro *et al.* 1992). *Mycoplasmas* were also isolated in sick animals from the Wadden Sea and are thought to be the causative organism of seal finger (Baker *et al.* 1998).

More recently *Brucella maris* has been isolated in seals and cetaceans from the North sea (Patterson *et al.* 1998). Bacteriological investigations have shown these organisms to be significantly different from other *Brucella* species. Serological studies of seals in particular have shown evidence of widespread infection in ten species of cetaceans and four species of seal. However, pathological changes associated with *B. maris* isolations have only been found in a total of nine cetacean and two seals, largely sub-clubber abscessation and pneumonia. A laboratory worker was infected with one isolate indicating that this is a potentially zoonotic agent (Patterson *et al.* 1998). However, in 1999 *Brucella* inducing abortions in bottlenose dolphins was reported. The causative organism was specific to this species and was named *Brucella delphini* (Miller *et al.* 1999). It is still not known how these two isolates are related or if they are the same organism. This bacterium appears to be quite widespread worldwide (Maratea *et al.* 2003).

Leptospira pomona has also been found in some marine mammals but has not been reported in those from UK waters. However recent preliminary research has found the occurrence of a different serotype in UK seals but it is not clear yet if this is a novel serotype (SMRU and Institute of Zoology, unpublished data). Leptospire can be highly pathogenic and have been associated with episodic outbreaks among California sea lions in which it causes abortion (Buck and Spotte, 1986; Colegrove, *et al.* 2005; Gulland *et al.* 1996).

Table 8. Viruses in marine mammals – From Visser *et al.* (1991).

Virus Family	Virus	Species
Adenoviridae	Sea Lion Hepatitis Virus	California sea lion Sei whale
Herpesviridae	Alphaherpesvirinae Phocine herpesvirus-1 Uncharacterised herpesvirus	Harbour seal California sea lion Beluga whale Harbour porpoise
Poxviridae	Seal poxvirus Parapoxvirus Orthopoxvirus	Harbour seal Grey seal California sea lion Northern fur seal S. American sea lion Bottlenose dolphin White sided dolphin Harbour porpoise Grey seal
Picornaviridae	Picornavirus	Harbour seal Grey whale
Caliciviridae	San Miguel sea lion virus Calicivirus	California sea lion Northern fur seal Northern elephant seal Pacific walrus Steller sea lion Grey seal Bottlenose dolphin Fin whale Grey whale Bowhead whale Sperm whale

Orthomyxoviridae	Influenzavirinae H7N7 Influenza A virus H4N5 H13N9 H13N2	Harbour seal Pilot whale Striped dolphin
Paramyxoviridae	Canine Distemper Virus (CDV) Phocine Distemper Virus (PDV) Porpoise Morbillivirus Dolphin Morbillivirus	Crabeater seal Baikal seal Harbour seal Grey seal Ringed seal Harp seal Harbour porpoise Striped dolphin
Coronaviridae	Coronavirus	Harbour seal
Rhabdoviridae	Rabies virus	Ringed seal
Retroviridae	Spumavirus	California sea lion
Papovaviridae	Papillomavirus	Burmeister's porpoise Cetacean spp.

Tuberculosis (*Mycobacterium tuberculosis*) has been diagnosed in various fur seal and sea lion species, (Cousins *et al.* 1990; Forshaw and Phelps, 1991; Bastida, 1999). Cousins *et al.* (2003) compared isolates from seals (pinnipeds) in Australia, Argentina, Uruguay, Great Britain and New Zealand to determine their relationships to each other. The seal isolates could be distinguished from other members of the *M. tuberculosis* complex on the basis of host preference and phenotypic and genetic tests. Pinnipeds appear to be the natural host for this 'seal bacillus', although the organism is also pathogenic in guinea pigs, rabbits, humans and possibly cattle. Cases of disseminated disease have been found. As with other members of the *M. tuberculosis* complex, aerosols are the most likely route of transmission. The name *Mycobacterium pinnipedii* sp. nov. has been proposed for this novel member of the *M. tuberculosis* complex.

Anthropogenic pathogens are largely found in marine mammals from the discharge of untreated sewage or effluent from facilities, which contain domestic animals. *Salmonella* species associated with man or his domestic animals have been cultured from marine mammals directly or their faeces, particularly *Salmonella bovis-morbificans* and *S. enteritidis* (Baker *et al.* 1995). In some cases these have been associated with pathologies and septicemia. It was found that between 1.4 and 11.8% of grey and harbour seals in the East coast of England taken into rehabilitation centres were positive for *Salmonella*. Although the origin of some of these organisms is not known, *S. bovis-morbificans* is generally specific to cattle and may indicate contamination of marine mammals by anthropogenic organisms.

3.3.3.3 Toxic Algae (Harmful Algal Blooms)

There have been a number of incidents in the US, and on the west coast of Africa, where toxins produced by algae have been associated with mortalities of marine mammals. Indeed such blooms appear to be regular and repeating events, causing mass mortalities of dolphins, sea lions and manatees (Hallegraeff, 1993; Flewelling *et al.* 2005). Unusual mortality events include dinoflagellate toxins in Florida manatees and Humpback whales (Geraci *et al.* 1989; O'Shea *et al.* 1991), brevetoxins in Bottlenose dolphins (Geraci, 1989; Flewelling *et al.* 2005), saxitoxin in sea otters (DeGange and Vacca, 1989), and ciguatoxin in Hawaiian monk seals (Gilmartin *et al.* 1987). Mass mortalities among California sea lions, linked to *Pseudo-nitzschia australis* that produces domoic acid, a neurotoxin found in fish and in the body fluids of the sea lions that died (Scholin *et al.* 2000) are also now a more regular occurrence.

3.3.4 Gaps in Knowledge

Whilst there has been a considerable amount of recent research on infectious and pathogenic diseases in marine mammals, particularly in the 10 years following the morbillivirus outbreaks of the 1980s and the 2002 PDV outbreak, we know surprising little about the occurrence and impact of other infections in European seal populations. Stranding schemes designed to determine mortality rates and the causes of

death of marine mammals around the UK have been forced by limited funding to concentrate their efforts on cetaceans rather than seals. Serological surveys could provide invaluable data on the exposure and immunity of populations to various diseases and this approach was proved useful in estimating the size of the susceptible harbour seal population in the UK before the recent outbreak of PDV in Europe (Thompson *et al.* 2002).

A small-scale survey of anthropogenic bacteria such as *Salmonella* has been conducted in seals but we have no information on the occurrence of anthropogenic viruses such as enteroviruses. Indeed some pilot work suggested that other sewage related organisms such as *Campylobacter* may be a risk for marine mammal health but this study has not been followed up. Recent pilot studies have found UK seals to have been widely exposed to leptospirosis and toxoplasmosis (SMRU, unpublished) therefore this type of baseline surveillance needs to be expanded.

4. BYCATCH AND OTHER NON-ENERGY-RELATED MANAGEMENT ISSUES

4.1 Bycatch

The accidental capture of marine mammals in fishing gear is an issue of some current concern throughout EU waters, and beyond. Work by the SMRU since 1993 has been targeted at determining accidental catch ('bycatch') rates of marine mammals in several fisheries in UK waters. Similar work has been conducted in many other European countries (SEC 2002) and is now mandated under EU Council Regulation 812/2004.

The SEA8 area and adjacent waters are exploited by fishing vessels from several EU and other states, and several of these fleets have been, or are currently being, monitored in order to quantify bycatch. Bycatch levels in this area are relatively high compared with some other SEA areas, due to the presence of large amounts of gillnetting, as well as significant levels of pelagic trawling during the winter, and relatively high densities of certain marine mammal species.

Gill and tangle net fisheries have been monitored sporadically since 1993 in UK waters, and in the early 1990s an estimated 2,200 porpoises annually were being caught in the UK and Irish Celtic Sea hake gillnet fishery alone. It is known that tangle net fisheries (for species such as monkfish, rays and spider crabs) also have a marine mammal bycatch, and although studies are underway to quantify the levels of such bycatch, no analysis has yet been completed.

Although the hake gillnet fishery has declined (in terms of the number of boats involved) since the 1990s, other static net fisheries in this region may have increased since then. The number of porpoises becoming caught in static nets in the region annually is therefore currently unknown, but is likely still in the hundreds at least. Both grey seals and common dolphins have also been recorded caught in these same sorts of fisheries, but there are no reliable estimates of the total level of bycatch for these species either.

The pelagic trawl fisheries of the region have been well-studied in recent years (Northridge *et al.* 2006). Porpoises and seals do not seem to be vulnerable to bycatch in these fisheries but common dolphins are relatively frequently taken and there are also a few records of other species such as Atlantic white-sided, Risso's and striped dolphins, and pilot whales in pelagic trawl fisheries of the wider region (Celtic Sea/Bay of Biscay). In the western channel common dolphins seem especially vulnerable to bycatch in pelagic pair trawls targeting bass in the winter months, when common dolphin densities are at a peak. Total mortalities in UK bass pair teams peaked at over 400 animals in the 2003-2004 winter and have since declined to less than 100 in 2005-2006.

Further estimates of marine mammal bycatch by species and by fishery are expected both for the UK and other neighbouring countries (Ireland, France and Spain, all of whose vessels fish in the SEA8 area) are expected in 2007-2008 in response to the obligations on member states of the EU under the bycatch regulation (812/2004).

4.2 Other issues

Another potential source of mortality to cetaceans may be through collisions with shipping. Whales are occasionally reported to be struck and killed, especially by fast-moving ferries, in other parts of the world, and smaller cetaceans can also be impacted by propeller strikes from small vessels. In some areas, where

ships are numerous and cetacean numbers are depleted, this can be a serious cause for concern. There are very few data with which to estimate the frequency of such events, and consequently this has not been identified as a significant source of additional mortality in this region, although the SEA8 area contains some very high densities of commercial and recreational shipping traffic, and is also an area frequented by large numbers of marine mammal individuals.

5. CONSERVATION FRAMEWORKS

5.1 Cetaceans

5.1.1 Europe

All cetacean species are listed in Annex IV (Animal and Plant Species of Community Interest in Need of Strict Protection) of the European Commission's Habitats Directive. Under Annex IV, the keeping, sale or exchange of such species is banned as well as deliberate capture, killing or disturbance.

The harbour porpoise and the bottlenose dolphin are also listed in Annex II of the Habitats Directive. Member countries of the EU are required to consider the establishment of Special Areas of Conservation (SACs) for Annex II species. SACs have been established for the bottlenose dolphin, one in the Moray Firth, Scotland and two in Cardigan Bay, Wales. No SACs have yet been established for the harbour porpoise.

The Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS) was formulated in 1992 and nine European countries including the UK are now Parties to the Agreement. Under the Agreement, provision is made for protection of specific areas, monitoring, research, information exchange, pollution control and heightening public awareness. Measures cover the monitoring of fisheries interactions and disturbance, resolutions for the reduction of by-catches in fishing operations, and recommendations for the establishment of specific protected areas for cetaceans. The UK applies the provisions of ASCOBANS to waters under its jurisdiction.

All cetacean species are listed on Annex A of EU Council Regulation 338/97 and are therefore treated by the EU as if they were on CITES Appendix I, thus prohibiting commercial trade.

5.1.2 UK

In British waters, all species of cetacean are protected under the Wildlife and Countryside Act 1981 and the Wildlife (Northern Ireland) Order 1985. Whaling is illegal under the Fisheries Act 1981.

Guidelines to minimise the effects of acoustic disturbance from seismic surveys, agreed with the oil and gas industry, were published by the then Department of the Environment in 1995 and are revised regularly. Member companies of UK Oil & Gas have indicated that they will comply with these Guidelines in all areas of the UK Continental Shelf. Under the Guidelines there is a requirement for visual and acoustic surveys of the area prior to seismic testing to determine if cetaceans are in the vicinity, and a slow and progressive build-up of sound to enable animals to move away from the source.

In 1999, the then Department of the Environment, Transport and the Regions produced two sets of guidelines aimed at minimising disturbance to cetaceans. The first, Minimising Disturbance to Cetaceans from Whale Watching Operations, is aimed at tour operators and members of the public involved in whale, dolphin and porpoise watching activities. The second, Minimising Disturbance to Cetaceans from Recreation at Sea, is aimed at anyone involved in any recreational activity in UK coastal waters who may incidentally encounter cetaceans.

5.2 Seals

5.2.1 Europe

The grey and harbour seal are listed in Annex II of the Habitats Directive under which member countries of the EU are required to consider the establishment of Special Areas of Conservation (SACs). A number of terrestrial candidate SACs have been established for grey and harbour seals around the coast of the UK. There are currently no marine candidate SACs.

All seal species are listed on Annex A of EU Council Regulation 338/97 and are therefore treated by the EU as if they were on CITES Appendix I, thus prohibiting commercial trade.

5.2.2 UK

Under the Conservation of Seals Act 1970, grey and harbour seals in the vicinity of fishing nets can be killed to prevent damage to the nets or to fish in the nets. Both species are protected during the breeding season: September-December in the case of grey seals; June-August in the case of harbour seals. However, licences to kill seals may be granted for any time of the year for specific listed purposes.

Under the Act, the Natural Environment Research Council (NERC) has a duty to provide scientific advice to government on matters related to the management of seal populations. NERC has appointed a Special Committee on Seals (SCOS) to formulate this advice so that it may discharge this statutory duty. Formal advice is given annually based on the latest scientific information provided to SCOS by SMRU. SMRU also provides to government scientific review of applications for licences to shoot seals, and information and advice in response to parliamentary questions and correspondence.

6. CONCLUSIONS

- The SEA8 area is an important area for some cetacean species. Harbour porpoises are very abundant in summer at least, and common dolphins are very abundant year round and especially in winter. Bottlenose dolphins and minke whales are relatively common in summer at least. Long-finned pilot whales and Risso's dolphins also occur regularly.
- These species are thus important predators in this region. Because of the link between the abundance and availability of fish prey and the reproductive success of marine mammals, changes in the availability of principal forage fish may be expected to result in population level changes of marine mammals. It is currently not possible to predict the extent of this.
- Seals are sensitive to the low frequency sounds generated by oil exploration and production but seal densities are low in the SEA8 area. Small cetaceans are relatively insensitive to low frequencies. Circumstantial evidence suggests that large whales may have good low frequency hearing.
- There are no reliable data to suggest that vessel noise or drilling noise adversely affect seals or small cetaceans.
- A relatively new source of noise in UK coastal waters is that associated with the construction and running of offshore wind farms. To date there is limited information on the noise generated during each of the survey, construction and operation phases. Harbour porpoises have shown equivocal responses to construction activity; harbour seals and harbour porpoises have shown relatively mild aversive behavioural responses to the playback of underwater noise from a simulated 2MW wind turbine.
- Contaminants, such as polychlorinated biphenyls, DDTs and chlorinated pesticides probably have toxic effects on the reproductive and immune systems of marine mammals. There is little evidence that heavy metals cause substantial toxic responses, except at high concentrations. Cetacean species which feed lower down the food chain may be at risk from exposure to polyaromatic hydrocarbons, although very little is known about current exposure levels or the effects of chronic exposure in marine mammals.
- Major oil spills are likely to result in direct mortality. More generally, marine mammals are less vulnerable than seabirds to fouling by oil, but they are at risk from chemicals evaporating from the surface of an oil slick at sea within the first few days. Individuals may drown as a result of associated symptoms. Neonatal seal pups are at risk from oil coming ashore.
- Bycatch levels in the SEA8 area are relatively high compared with other SEA areas, due to the presence of large amounts of gillnetting, significant levels of pelagic trawling during the winter and high densities of harbour porpoise and common dolphin.

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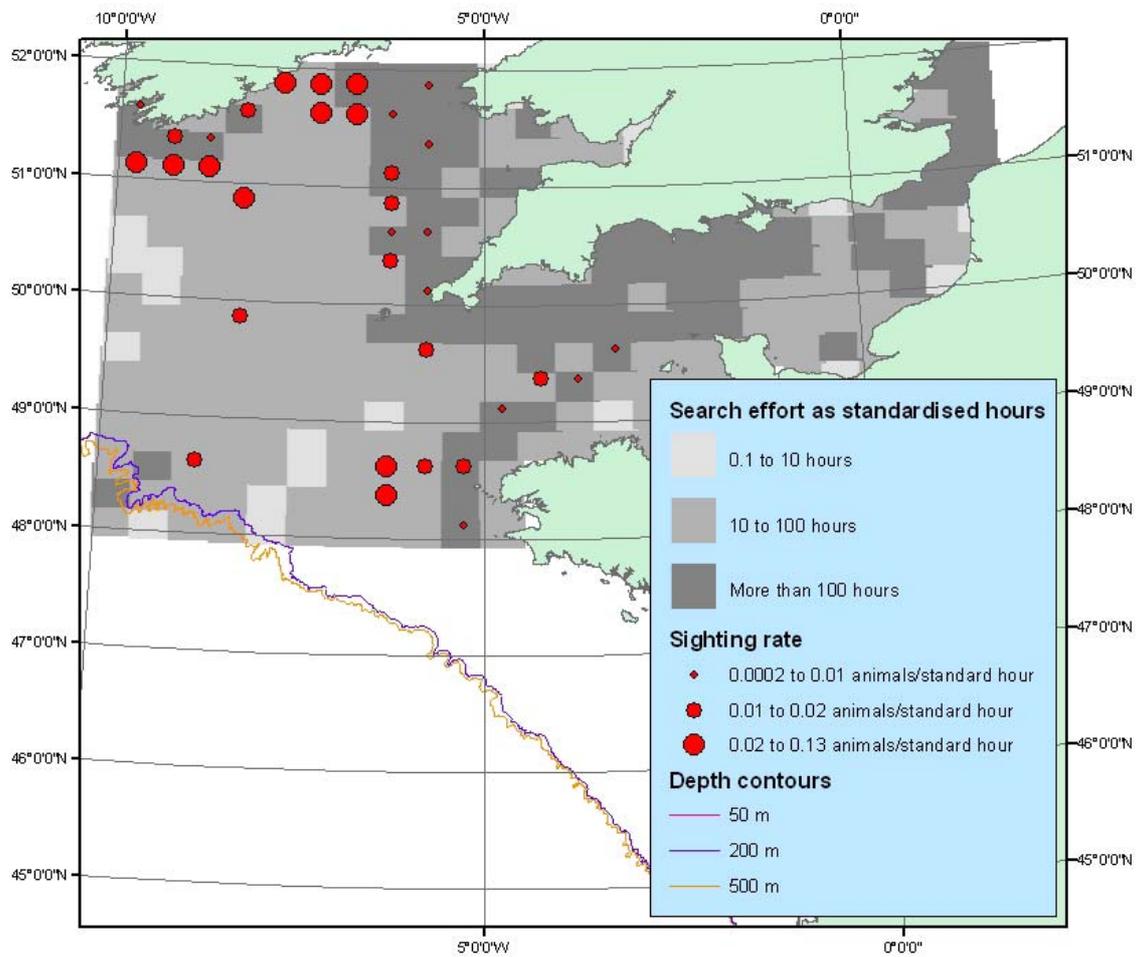


Figure 1 – Distribution of minke whale (*Balaenoptera acutorostrata*) to the south and west of the UK (adapted from Reid *et al.* 2003)

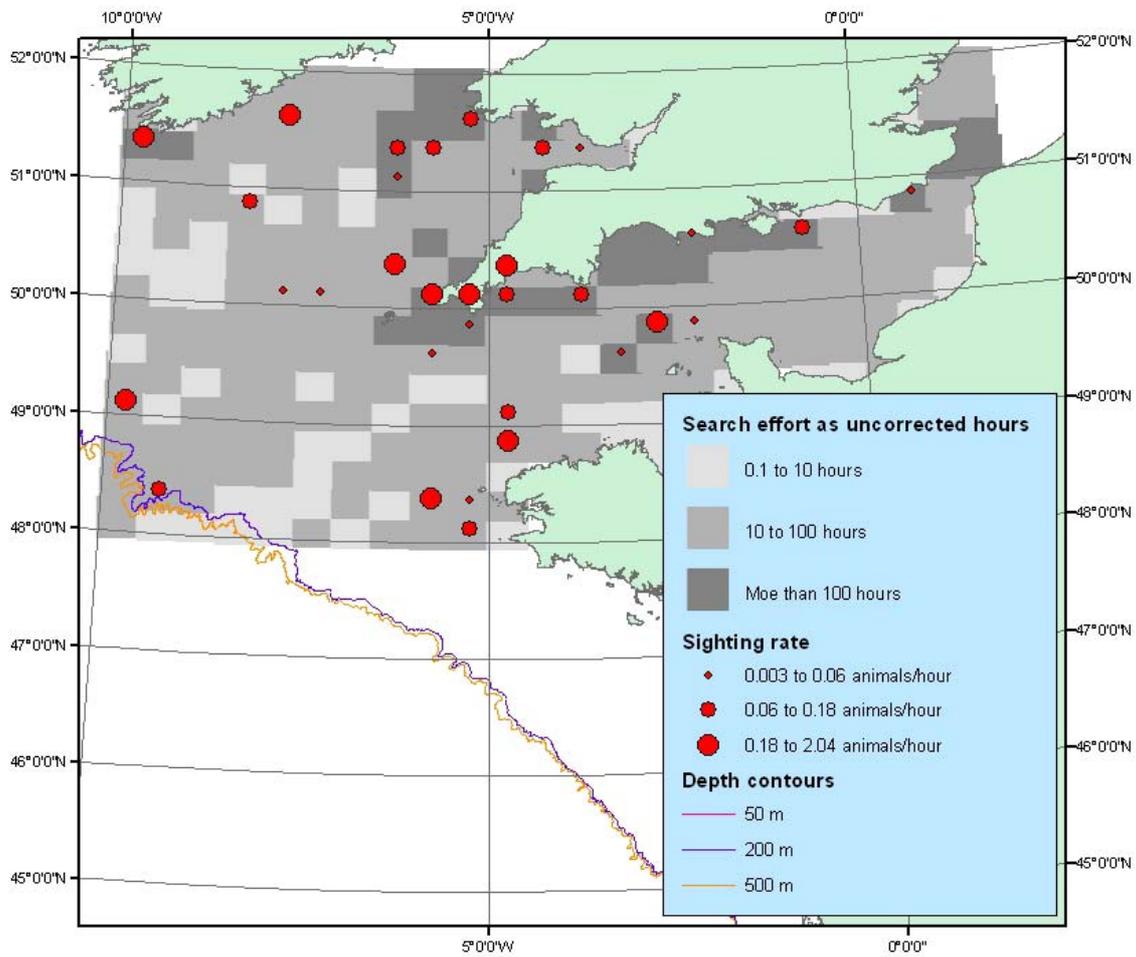


Figure 2 – Distribution of common bottlenose dolphin (*Tursiops truncatus*) to the south and west of the UK (adapted from Reid *et al.* 2003)

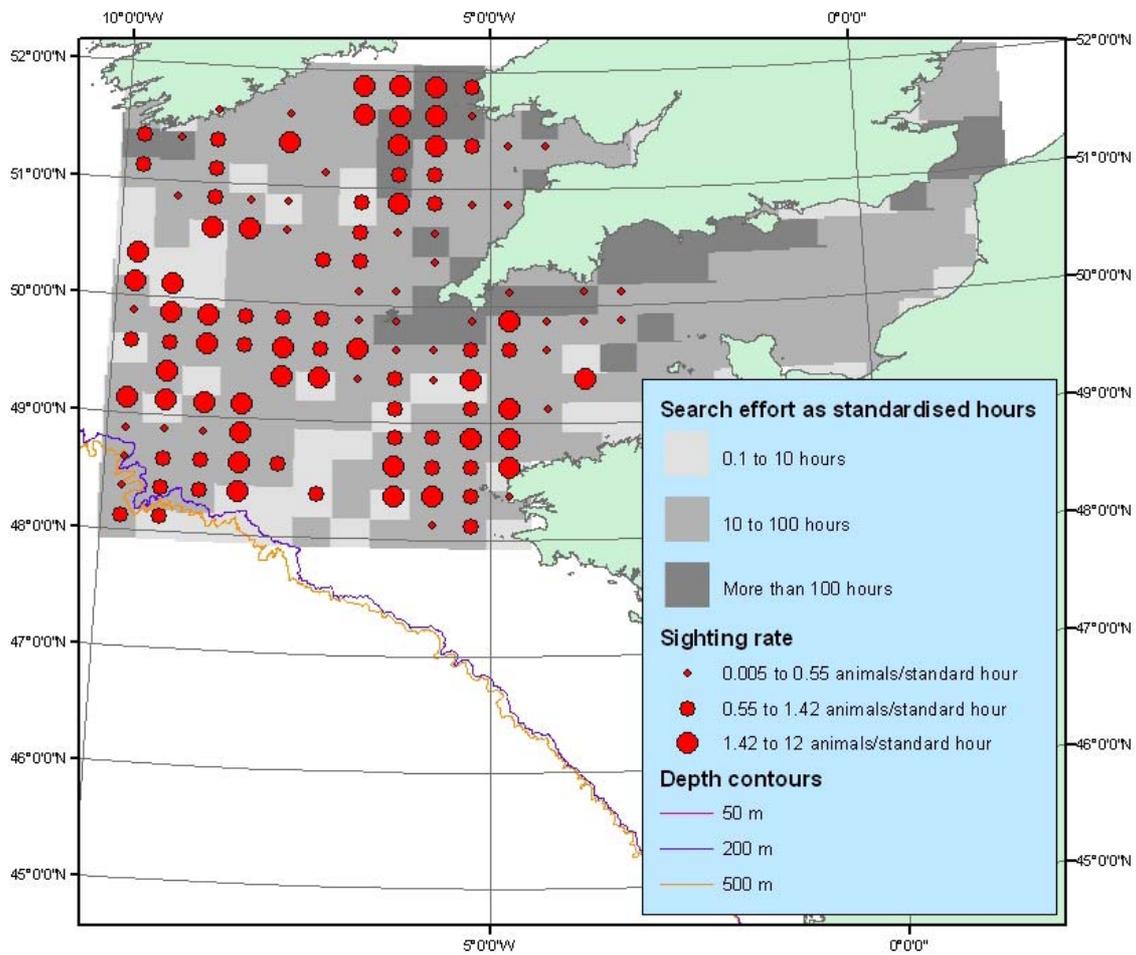


Figure 3 – Distribution of short-beaked common dolphin (*Delphinus delphis*) to the south and west of the UK (adapted from Reid *et al.* 2003)

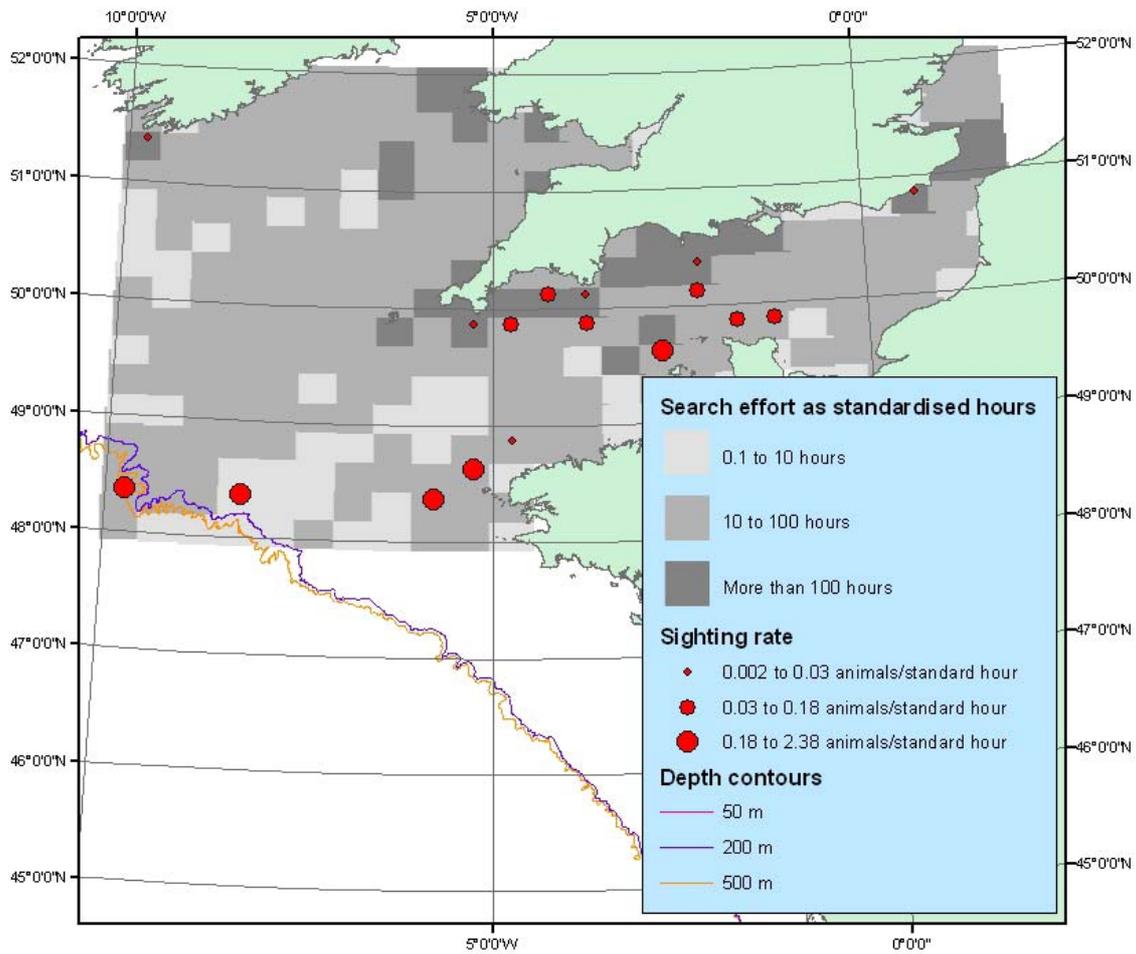


Figure 4 – Distribution of long-finned pilot whale (*Globicephala melaena*) to the south and west of the UK (adapted from Reid *et al.* 2003)

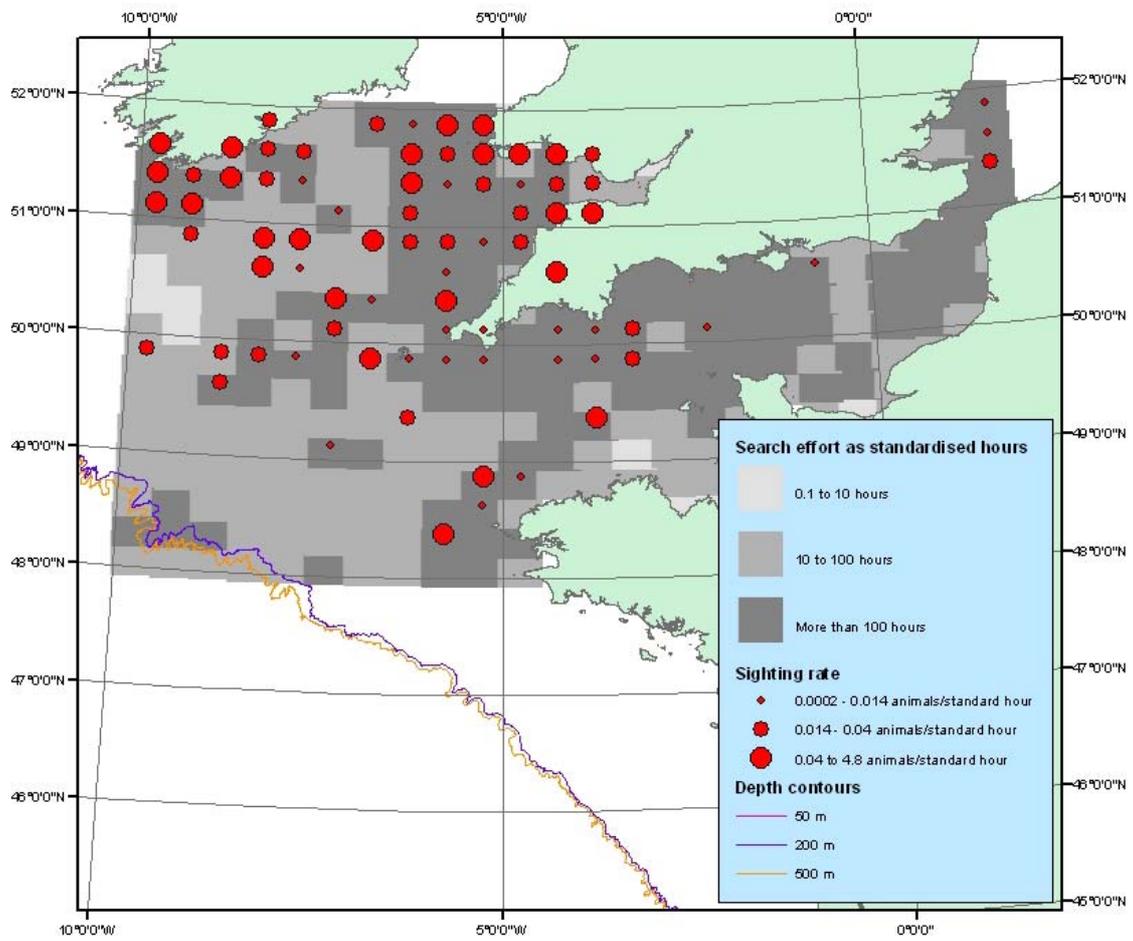


Figure 5 – Distribution of harbour porpoise (*Phocoena phocoena*) to the south and west of the UK (adapted from Reid *et al.* 2003)

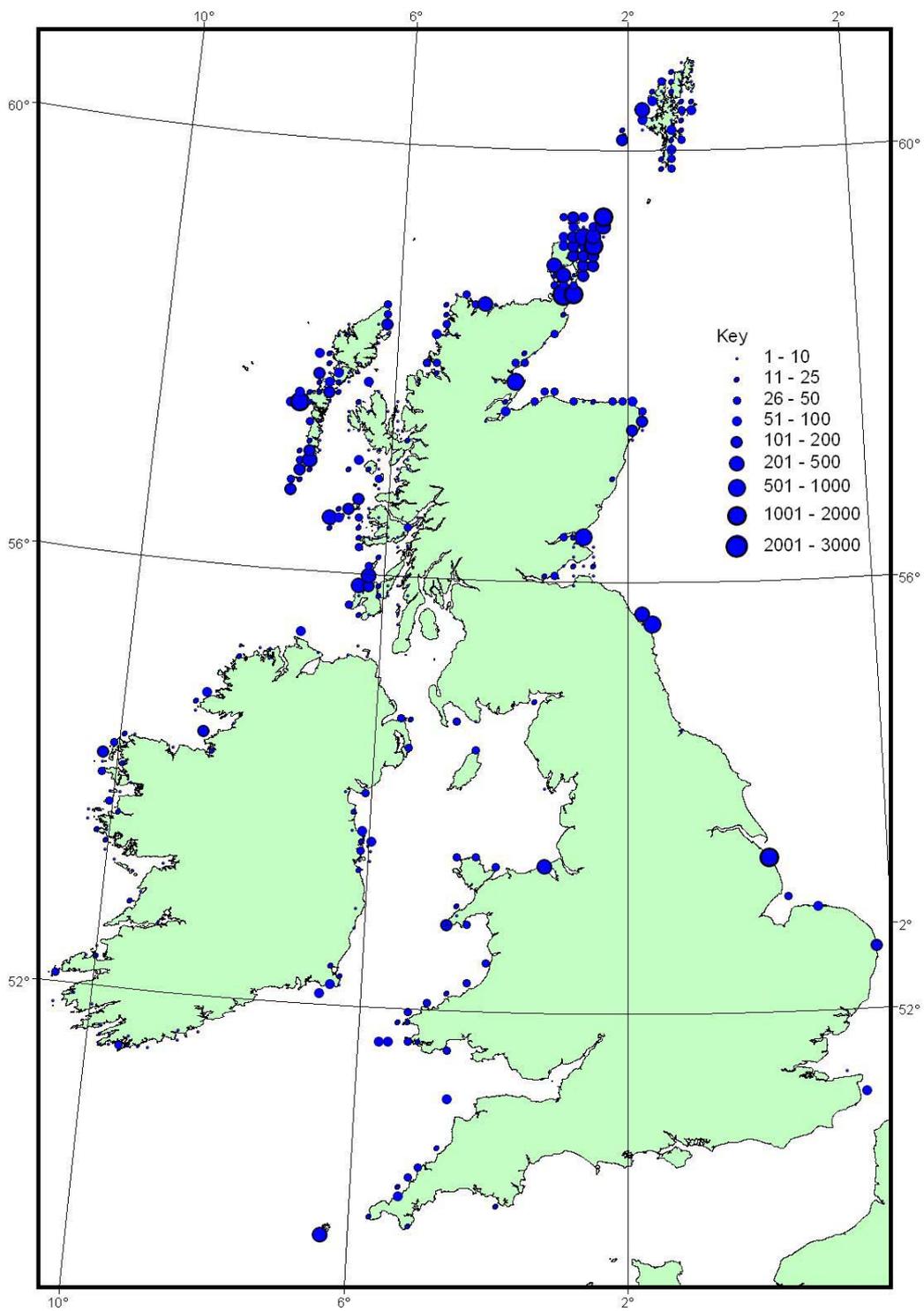


Figure 6. Distribution of grey seals at haul-out sites around Britain and Ireland in August.

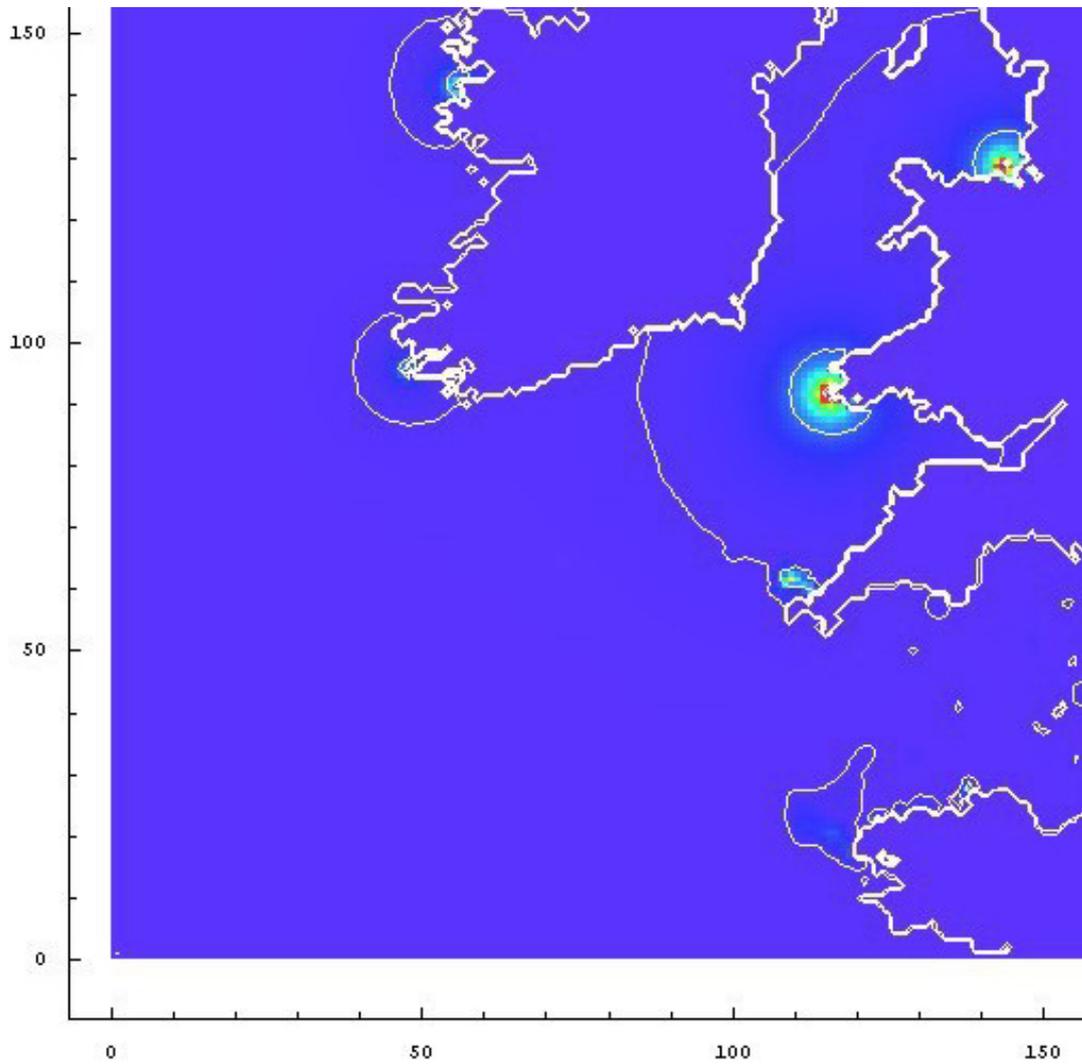


Figure 7. Estimated at-sea usage by grey seals in the SEA8 area (from Matthiopoulos *et al.* 2004), including data from seals tagged in Brittany but excluding data from seals tagged in Wales. Warm colours indicate higher usage.

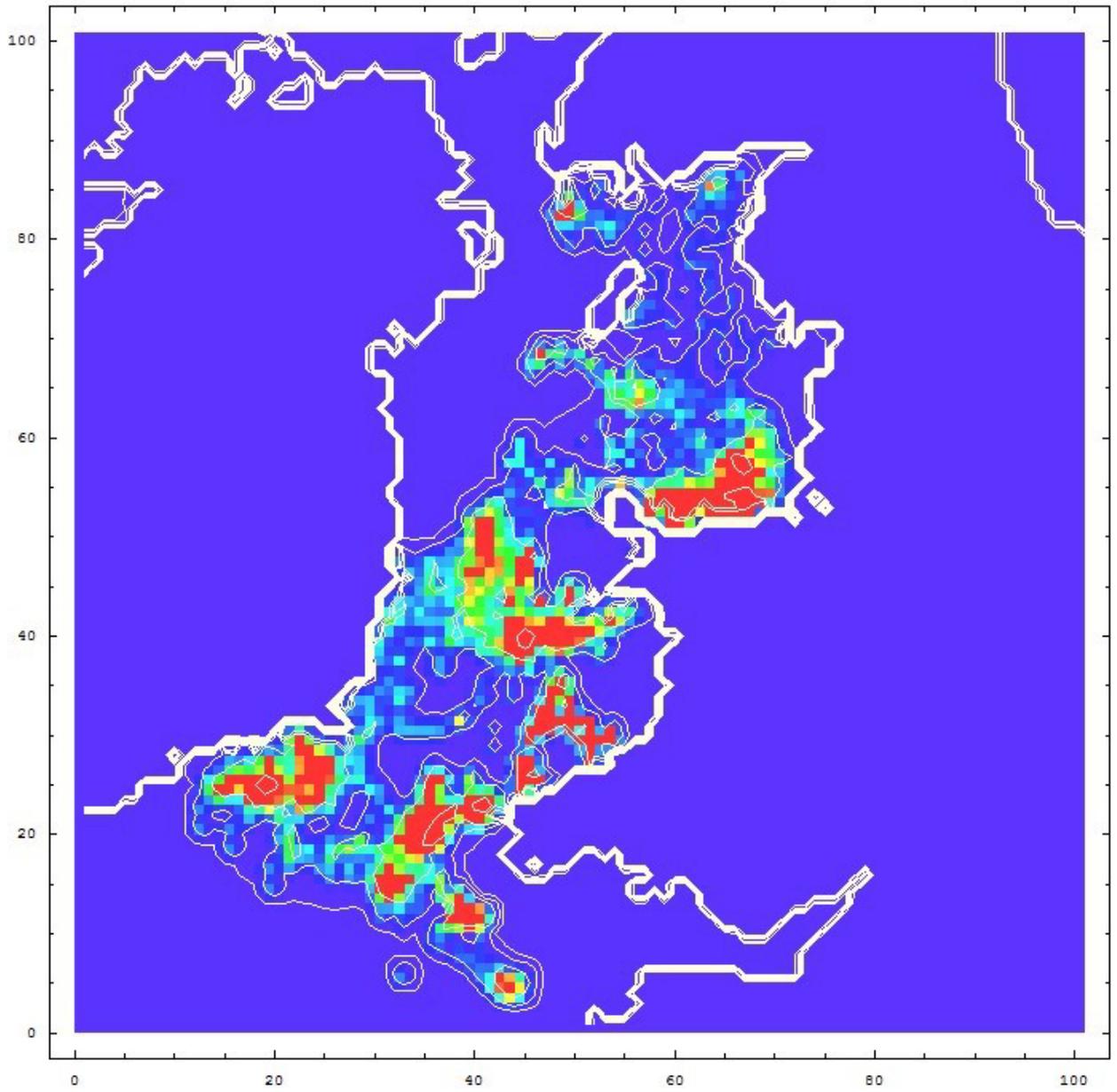


Figure 8. Estimated at-sea usage by grey seals tagged in Wales in 2004. Warm colours indicate higher usage.

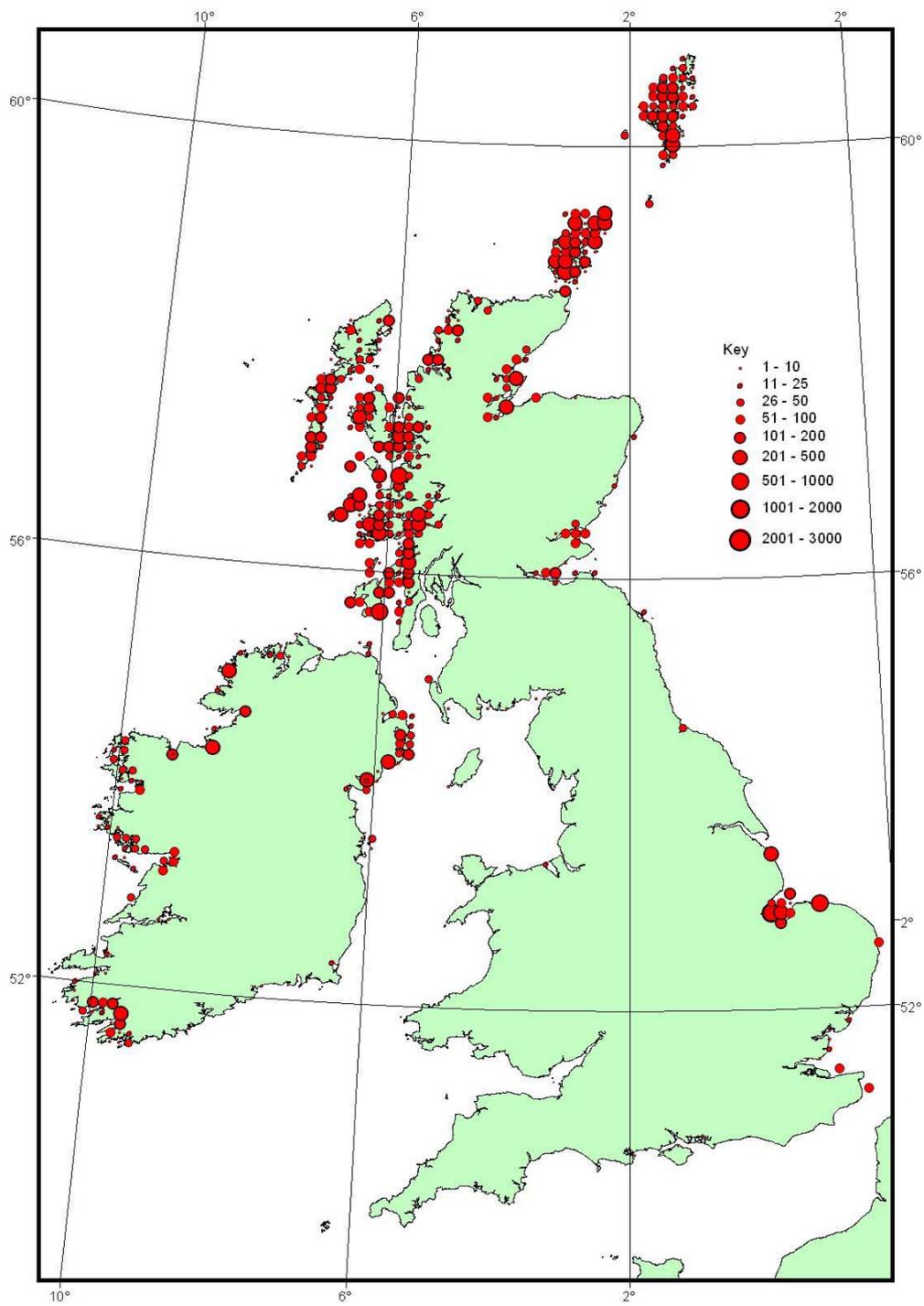


Figure 9. Distribution of harbour seals at haul-out sites around Britain and Ireland in August.