

THE SHORT-BEAKED COMMON DOLPHIN (*DELPHINUS DELPHIS*) IN THE NORTH-EAST ATLANTIC: DISTRIBUTION, ECOLOGY, MANAGEMENT AND CONSERVATION STATUS

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The common dolphin is the second most abundant cetacean species in the North-east Atlantic, with a wide-ranging distribution and is, potentially, impacted by a wide variety of pressures and threats. To assess the conservation status of common dolphins in this region, it is essential to understand population structure, key drivers of population dynamics, key resources and the effects of stressors. In recent years, a number of studies have assessed population structure, distribution and abundance, life-history parameters, dietary requirements and the effect of stressors—especially those caused by anthropogenic interactions, such as incidental capture (i.e., by-catch) and pollutants. A full review of this work is presented, with particular focus on current and potential pressures and threats. Notwithstanding the recent research, due to the lack of baseline data (i.e., prior to human influence) on abundance and pregnancy rate and on historical direct and incidental capture rates, the actual conservation status of the North-east Atlantic common dolphin population is unknown. Current assessments of conservation status of the species are therefore reliant on recent data. However, these assessments are hindered by the lack of data on contemporary incidental capture rates in some fisheries and limited sampling in other fisheries, as well as large data gaps for other stressors. In addition, the numerous potential ways in which multiple and diverse stressors can interact remain poorly understood. This chapter provides an outline of a management framework and describes methods for future evaluation of conservation status through development of indicators focusing on not only population size and distribution but also mortality and condition. Recommendations for research and conservation actions are described.

Introduction

Common dolphins (*Delphinus delphis*) are one of the most abundant cetaceans in the North-east (NE) Atlantic and are potentially impacted by a wide variety of threats and pressures. A large number of studies since the year 2000 have focused on their biology, ecology, population structure, abundance, health status, foraging behaviour, interactions with fisheries, and pollutant levels,

among other aspects. No baseline data (prior to anthropogenic impacts) are available for this species in the region, making comparisons to the modern situation impossible. Current assessments of conservation status of the species therefore rely on recent data. Such assessments require information on genetic diversity, evidence of density-dependent compensatory responses in reproductive parameters, and, most importantly, trend analysis of abundance estimates—though for the last only primary (preliminary) survey data are available for most of the NE Atlantic. As these parameters respond to a change after it has occurred, most likely years later, monitoring of anthropogenic (and environmental) impacts is extremely important to limit their effects. Under European Union legislation, Member States now have a legal obligation to undertake such monitoring in their national waters. The present review encompasses a comprehensive assessment of knowledge on the common dolphin in the NE Atlantic, with the aim of providing both a current evaluation of conservation status and valuable information for the future development of a conservation management plan in these waters.

Species identification

The common dolphin was first identified by Artedi in 1738 and later described by Linnaeus in 1758. Much confusion has arisen about whether the ‘common dolphin’ comprises one or more species due to the cosmopolitan distribution and high variability in morphological characters and pigmentation patterns of otherwise-similar forms. Almost two dozen nominal species have been described (Hershkovitz 1966, Heyning & Perrin 1994).

During the 1990s, studies focused on clarifying taxonomic status and proposed that only two distinct species of common dolphin existed: the short-beaked common dolphin *Delphinus delphis* Linnaeus, 1758 and the long-beaked common dolphin *D. capensis* Gray, 1828 (Heyning & Perrin 1994, Rosel et al. 1994). Contemporary taxonomic classifications further recognize an endemic subspecies of the short-beaked form, *D. delphis ponticus* (Barabash-Nikiforov, 1935), which is restricted to the Black Sea, and two endemic subspecies of the long-beaked form found in the Indo-Pacific region: *D. capensis capensis* Gray, 1828, and the extremely long-beaked *D. c. tropicalis* (van Bree, 1971) (Amaha 1994, Heyning & Perrin 1994, Rosel et al. 1994, Jefferson & Van Waerebeek 2002, Natoli et al. 2006, Perrin 2009).

Delphinus delphis and *D. capensis* were differentiated on the basis of an assessment of morphological data (including measurements of skeletal characters, such as overall body size, length of the rostrum, rostrum length/zygomatic width ratio [RL/ZGW], and tooth counts) and colouration patterns in two sympatric populations inhabiting coastal waters off California, United States (Heyning & Perrin 1994). An assessment of samples and data from outside this region led the authors to propose that *D. delphis* and *D. capensis* existed globally as two separate species. This view was further supported by a parallel genetic analysis of tissue samples from dolphins inhabiting Californian waters and from *D. delphis* in the eastern tropical Pacific (ETP) and Black Sea. No shared mitochondrial DNA (mtDNA) haplotypes or cytochrome *b* sequences were observed between the two morphotypes, and both gene regions exhibited nucleotide frequency differences and fixed nucleotide substitutions between the two morphotypes (Rosel et al. 1994). The estimated genetic divergence in the mitochondrial DNA control region between the two forms was 1.1% (Rosel et al. 1994).

However, the classification of individuals into the two species was not as clear cut as originally perceived, and questions have arisen over the use of morphology-based taxonomy in common dolphins. Additional morphological studies have shown large variability in body and skull size and RL/ZGW ratio in *D. delphis* inhabiting the North Atlantic and waters around New Zealand and southern Australia compared with *D. delphis* off the coast of California (Bell et al. 2002, Murphy et al. 2006, Westgate 2007, Jordan 2012). Studies using mtDNA identified that *D. delphis* and *D. capensis* do not show reciprocal monophyly (LeDuc et al. 1999, Amaral et al. 2007), and an assessment of nuclear divergence using amplified fragment length polymorphism markers

suggested that *D. delphis* and *D. capensis* diverged only recently (Kingston & Rosel 2004). Overall, the short-beaked form, despite having high morphological variability, exhibits low genetic differentiation between populations, with evidence of gene flow across oceans, reflecting high mobility and a fluid social structure in this species (Natoli et al. 2006; assessed nine microsatellite and 369 base pairs [bp] of the mtDNA control region). The opposite is true for the long-beaked form: populations are highly differentiated, suggesting that separate populations may have evolved from independent founder events, then converged on the same morphotype (Natoli et al. 2006). A more recent international collaborative genetics study, which analysed sequences of the mitochondrial DNA cytochrome *b* gene, further revealed that the distribution of mitochondrial lineages does not correspond to the geographical distribution of the long-beaked morphotype, thus suggesting some ambiguity in the phylogenetic relationships and taxonomy within this species (Amaral et al. 2009, 2012a,b). Further analysis is required to clarify the species-level taxonomy of common dolphins (International Whaling Commission [IWC] 2009).

North-east Atlantic Ocean

Only the short-beaked form has been recorded in the North Atlantic (Murphy et al. 2006, Westgate 2007). Based on tooth counts, absolute length of rostrum and RL/GZW ratio, Murphy et al. (2006) proposed that common dolphins in the NE Atlantic are most similar to *Delphinus delphis* described by Heyning and Perrin (1994) but are larger than *D. delphis* in waters off California. The ranges of RL/GZW ratio defined by Heyning and Perrin (1994) for *D. delphis* and *D. capensis* were 1.21–1.47 and 1.55–1.77, respectively. Skulls of mature common dolphins from the NE Atlantic had RL/GZW ratios in the range of 1.31 to 1.57 (mean = 1.41, $n = 111$), with 95% of ratio values less than 1.52 (Murphy et al. 2006). Upper alveolar count and rostrum lengths had ranges of 41–56 and 233.6–299.8 mm, respectively, compared with 42–54 and 218–275 mm for California *D. delphis* (Heyning & Perrin 1994). In Mauritanian waters, morphological analysis indicates that both short- and long-beaked forms were present (Van Waerebeek 1997, Pinela et al. 2011), although stable isotope analysis of diet indicated that skull variability in this region (e.g., RL/GZW ratio range of 1.27–1.76) may be due to niche segregation rather than speciation (Pinela et al. 2011).

Species identification in the NE Atlantic has also been confirmed by genetic analysis. Using mtDNA, Natoli et al. (2006) identified only *D. delphis* in waters from the Canary Islands to Scotland ($n = 100$), with the nearest *D. capensis* population reported off Mauritania ($n = 6$), which was highly differentiated from long-beaked dolphins off South Africa. Amaral et al. (2007) also identified only *D. delphis* in a sample of 69 individuals stranded in Portugal, northern Spain and Scotland; the identification was based on analysis of two mitochondrial regions (control region and cytochrome *b* gene). Interestingly, that study documented a group of highly divergent individuals, ‘Clade X’ (five females throughout the sampled area). Genetic divergence between Clade X and *D. delphis* in the NE Atlantic was 1.59%, considerably higher than the divergence reported by Rosel et al. (1994) for *D. delphis* and *D. capensis*. Genetic divergence between Clade X and *D. capensis* was also high at 1.76% (Amaral et al. 2007). In a broader study that included populations from the Atlantic, Indian and Pacific Oceans, it became clear that Clade X was present not only in the North Atlantic short-beaked common dolphin, but also in *D. capensis tropicalis* of the Indian Ocean (Amaral et al. 2012b). The most likely explanation for such a divergent clade is a scenario of variance and secondary contact during the Pleistocene, a period when the Atlantic and Indo-Pacific basins were intermittently isolated. This explanation has been proposed for phylogeographic patterns in other large migratory marine animals (Amaral et al. 2012b).

Individuals resembling long-beaked common dolphins in colour pattern and rostrum length have been sighted off the Azores (Quérouil et al. 2010) but have not been genetically characterized.

Population structure in the North Atlantic Ocean

Common dolphins are widely distributed throughout the North Atlantic, occurring in many jurisdictions enforcing different environmental legislation. Knowledge of population status and range is essential for effective conservation and management of this species in the region, and this has provided the impetus for numerous population studies in recent years.

Geometric analysis of cranial variables revealed significant differences between the NE Atlantic and North-west (NW) Atlantic for both male and female short-beaked common dolphins inhabiting continental shelf and slope waters ($n = 149$ males; $n = 96$ females) (Westgate 2007). Rostral width dimensions were important discriminating variables, suggesting differing feeding strategies between the populations (Westgate 2007).

Using genetic data, Natoli et al. (2006) also found significant but low differentiation between *Delphinus delphis* from the NW ($n = 13$) and NE Atlantic ($n = 119$). Genetic differentiation was more marked in maternally inherited mtDNA markers than nuclear markers, which did not always indicate significant differentiation, suggesting a greater dispersal of males. However, analysis of mtDNA data using the MIGRATE program (Beerli 2012) indicated a possible bias in the long-term direction of migration of females in the North Atlantic, from west to east (Natoli et al. 2006). Following this study, Mirimin et al. (2009b) assessed genetic variability at the mtDNA control region (360 bp) and 14 microsatellite loci using a larger sample size of stranded common dolphins, and individuals incidentally caught in fishing gear, from continental shelf and slope waters of the NE ($n = 205$) and NW ($n = 219$) Atlantic. Results confirmed those of Natoli et al. (2006), and significant genetic differentiation was observed, which was more pronounced in mitochondrial than nuclear markers, suggesting the existence of at least two genetically distinct populations (Mirimin et al. 2009b). However, the low levels of genetic differentiation measured in both studies may arise from a recent population split or a high level of gene flow in the North Atlantic (Mirimin et al. 2009b). Natoli et al. (2006), Amaral et al. (2007) and Mirimin et al. (2009a) all reported highly significant negative values of a genetic index based on the distribution of alleles or haplotypes, Fu's F_s , indicating population expansion within the NE Atlantic. This index quantifies departures from the pattern of DNA polymorphism expected under a neutral model of evolution, caused, for example, by rapid population increase. A separate common dolphin population is recognized within the Mediterranean Sea based on mtDNA markers (Natoli et al. 2008).

Even in light of these recent studies, a full assessment of population structure in the North Atlantic has been hindered by sampling limitations as all genetic samples were obtained from continental shelf and contiguous oceanic waters. Further investigations into whether common dolphins from the NE and NW Atlantic belong to the same population or to two separate, but highly connected, populations requires sampling from the entire range of this species in the North Atlantic (International Council for the Exploration of the Sea [ICES] Working Group on Marine Mammal Ecology [WGMME] 2009, Murphy et al. 2009a).

North-east Atlantic Ocean

There appears to be a slight latitudinal cline in size of male common dolphins in the NE Atlantic, with males in higher latitudes slightly larger in total body length, absolute skull width, mandible length and depth compared to individuals off the NW coast of Spain (Murphy et al. 2006). Analysis of cranial morphometrics has also revealed evidence of population differentiation, with female *Delphinus delphis* off Portugal showing segregation from more northerly sampled areas (Murphy et al. 2006). Although samples of mature individuals from southern regions were small in the study by Murphy et al. (2006), there were indications that females off the Portuguese coast (the most southerly sampled region) may be mixing with individuals in the Mediterranean Sea population or individuals from waters further south of the sampled region.

Morphological traits are influenced heavily by environmental factors and should be interpreted differently from genetic traits. However, the results of the cranial morphometric study were partially supported by genetic analysis as a directional movement of females from the western Mediterranean Sea (Alborán Sea) into the North Atlantic was reported (Natoli et al. 2008). Based on genetic analysis of nine microsatellites, no significant genetic differentiation was detected in common dolphins sampled in the Alborán Sea ($n = 34$) and the contiguous Atlantic, though mtDNA analysis (426 bp) indicated significant differentiation (Galicia, $n = 30$; Portugal, $n = 17$).

Viricel (2006) did not observe any significant genetic variation using mtDNA (933 bp) and (7–11) microsatellite nuclear loci markers in samples obtained from oceanic ($n = 14$) and neritic ($n = 106$) waters of the Bay of Biscay and the English Channel ($n = 48$). Both Natoli et al. (2006) and Amaral et al. (2007) extended the sampling region in the NE Atlantic for assessing population structure, but using smaller sample sizes. Natoli et al. (2006) reported low but significant genetic differentiation between common dolphins in Scottish waters ($n = 26$) and those from the Celtic Sea ($n = 41$) and Galicia (NW Spain, $n = 39$) using nine microsatellites, but not sequences from the maternally inherited mtDNA control region. In contrast, Amaral et al. (2007) detected no significant genetic structure using mtDNA cytochrome *b* sequence data in samples from Scotland, Galicia and Portugal, except when the sexes were analysed separately, but sample sizes were too small for further assessment.

Following these initial investigations, Mirimin et al. (2009a) undertook a large-scale study, incorporating samples and data produced by Viricel (2006) and Amaral et al. (2007). In this study, 25 microsatellites and 556 bp of the control region were analysed in common dolphins from Scotland ($n = 13$), Ireland ($n = 102$), Celtic Sea ($n = 75$), English Channel ($n = 2$), France ($n = 46$), and Portugal ($n = 39$). Results indicated strong gene flow and the presence of a large ‘coastal’ panmictic (random mating) population in the NE Atlantic (Mirimin et al. 2009a). The Scottish sample demonstrated a unimodal distribution of the observed number of differences between pairs of haplotypes, suggesting the population has passed through a recent demographic expansion, but not a significant, negative value of F_{ST} , which suggests that the marginal position in the geographic range may have led to a lower genetic exchange rate with neighbouring aggregations (Mirimin et al. 2009a). Conversely, this was not the case for the Portuguese sample.

Moura (2010) assessed whether genetic differentiation within the NE Atlantic common dolphin population increases with geographic distance, that is, isolation by distance, which has been reported in the harbour porpoise, *Phocoena phocoena*, population in this region (Fontaine et al. 2007). Analysis of 15 microsatellite DNA markers in 466 samples from *Delphinus delphis*, inhabiting waters off Scotland to Madeira, showed a lack of population genetic structure along the European Atlantic coastline and a lack of evidence of isolation by distance. In addition, analysis of a large number of (Atlantic) Iberian common dolphins by Moura (2010) did not support the previous suggestion by Amaral et al. (2007) of fine-scale population structure in that region.

In more southern waters, analysis using mtDNA and 14 microsatellite loci did not detect evidence of genetic structure among common dolphins inhabiting waters off Madeira ($n = 56$) and the archipelago of the Azores ($n = 91$), which are 900 km apart (Quérrouil et al. 2010). Further evidence of gene flow over large geographic distances was provided by a lack of significant genetic differentiation amongst pairwise comparisons using microsatellites and samples from the central-east (CE) Atlantic (Azores, Madeira and Canary Islands, $n = 13$) and NE Atlantic (Galicia, Celtic Sea and Scotland, $n = 106$; Natoli et al. 2006). Shared haplotypes were also common between both regions, though sample sizes were small. Although F_{ST} values indicated no significant difference between the CE and NE Atlantic, ϕ_{ST} values indicated a significant difference between samples from the CE Atlantic and those from Scotland and Galicia. Amaral et al. (2012a) reported genetic differentiation between putative populations in the NE Atlantic (Portugal, northern Spain, Ireland and Scotland, $n = 75$) and the CE Atlantic (Madeira, $n = 29$) (and also the NW Atlantic, $n = 38$) using 14 microsatellites. As both studies reported low and hence marginally significant levels of divergence, this

suggests high levels of gene flow between the NE and CE Atlantic or a recent population split—a similar situation to that reported between the NE and NW Atlantic populations (Mirimin et al. 2009b).

Interestingly, unlike in the North Atlantic, fine-scale (≤ 1000 km) population genetic structure has been reported in *D. delphis* inhabiting the NE Pacific and waters around Australia and New Zealand (Möller et al. 2011, Amaral et al. 2012a). Oceanographic variables (such as ocean currents, chlorophyll concentrations, temperature and salinity) have been proposed as factors limiting the movements of common dolphins in these regions through their effects on local prey availability (Bilgmann et al. 2008, Möller et al. 2011, Amaral et al. 2012a). This was supported by a seascape genetics study that assessed total averages of chlorophyll concentration, water turbidity and sea-surface temperature (SST) for an 8-year-period and reported that marine productivity and SST were correlated with genetic structure in common dolphins at medium spatial scales, that is, within ocean basins, such as the North Atlantic and South Indo-Pacific (Amaral et al. 2012a).

In summary, morphometric and genetic assessments indicate that there is only one short-beaked common dolphin population in the NE Atlantic, ranging from Scotland to Portugal (Murphy et al. 2009a). Future studies should focus on sampling dolphins from offshore waters using remote biopsy darting systems and strategic sampling approaches (e.g., temporal [including seasonal] and spatial; ICES WGMME 2009, Mirimin et al. 2009a, Murphy et al. 2009a).

Ecological stocks

As a consequence of the low genetic differentiation in this species as a whole within the North Atlantic Ocean, it was proposed by the ASCOBANS/HELCOM* Small Cetacean Population Structure Workshop (Murphy et al. 2009a) and by the ICES WGMME (2009) that common dolphins in the NE Atlantic should be managed on an ecological timescale (i.e., managing ‘ecological stocks’). Ecological stocks can be defined as groups of individuals of the same species that co-occur in space and time and have an equal opportunity to interact with each other (Waples & Gaggiotti 2006). To take this approach, it was recommended that directed studies using ecological markers should be carried out to identify the existence of ecological stocks in common dolphins through sampling a large number of animals throughout the region, including all age-sex-maturity classes (ICES WGMME 2009, Murphy et al. 2009a). In addition, statistical power analysis should be undertaken to determine appropriate sample sizes required to detect the existence of ecological stocks within the NE Atlantic population.

The ASCOBANS/HELCOM Small Cetacean Population Structure Workshop discussed how to integrate different lines of evidence, including both genetic and ecological markers, to obtain the best possible indication of relevant stock structure. A few generations was proposed as the appropriate time frame for defining a management unit (MU) (Evans & Teilmann 2009). An MU was identified as a group of individuals for which there are different lines of complementary evidence suggesting reduced exchange (migration/dispersal) rates (Evans & Teilmann 2009). It was recommended that quantitative parameters (e.g., a maximum of 10% migration per generation) should be set, though in most cases this information is not available and the theoretical framework for integration of different evidence bases has not been fully developed (Evans & Teilmann 2009).

‘Elemental profiles’, determined, for example, by stable isotopes, fatty acids and pollutants, can be used as ‘ecological tracers’ of resource and habitat use (Caurant et al. 2009). However, these profiles can be sensitive to the physiological and health status of the animal and decomposition state of dead animals (Aguilar et al. 1999, Das et al. 2004, Pierce et al. 2008, ICES WGMME 2009),

* ASCOBANS: ‘Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas’; HELCOM: the Helsinki Commission, the governing body of the Convention on the Protection of the Marine Environment of the Baltic Sea Area.

as well as biotic factors such as metabolism, age, sex, and reproductive status (Evans & Teilmann 2009). For example, females transfer some of their lipophilic pollutants, such as polychlorinated biphenyls (PCBs), to their offspring during gestation and lactation, which confounds the use of these markers for assessing ecological stocks. Accumulation of certain xenobiotics is greater in females, possibly due to their lower capacity for detoxification compared to males (Aguilar et al. 1999 and references therein). This has been attributed to differing sex hormones and their effect on the activity of enzymes responsible for pollutant degradation (Aguilar et al. 1999). Stable isotope ratios indicated that female *Delphinus delphis* in the Southern California Bight consumed higher trophic level prey (higher $\delta^{15}\text{N}$ value) than males (Berman-Kowalewski & Newsome 2009, S.D. Newsome, personal communication, December 2012). However, $\delta^{15}\text{N}$ values will be dependent not only on an individual's food source and trophic level but also on its nutritional and physiological state (Cherel et al. 2005, Habran et al. 2010). All these confounding effects have to be considered prior to proposing ecological stocks within a population on the basis of elemental profiles (ICES WGMME 2009). In addition, toxicokinetics (toxicant absorption, distribution, metabolism, and excretion) and toxicodynamics (molecular, biochemical, and physiological effects of toxicants or their metabolites in biological systems) of some elements are only well known in humans and small laboratory mammals (Evans & Teilmann 2009).

Differing ecological tracers provide information on different timescales. For example, stomach contents provide information on dietary intake over a few days, fatty acids in blubber provide information on dietary intake a few weeks earlier (Nordstrom et al. 2008), stable isotopes in muscle provide information over a timescale of several months (Méndez-Fernandez et al. 2012), whereas stable isotopes in hard tissues, such as teeth (Graham et al. 2010), and cadmium in kidney (Lahaye et al. 2005) reflect dietary intake over years (Caurant et al. 2009). Again, studies based on these approaches have to take into account biotic factors, health status and condition, as well as temporal and seasonal variations in dietary intake and habitat use. Although one approach would be to assess numerous tracers to identify ecological stocks within the NE Atlantic population, the appropriate timeframe for defining an MU should be specified at the outset.

To date, the presence of ecological stocks has not been verified in the NE Atlantic common dolphin population owing to a lack of data (ICES WGMME 2009, 2012). Limited studies have been undertaken, with predominantly small sample sizes, thus increasing the possibility of sampling biases.

Das et al. (2003) reported that mean muscle and liver $\delta^{13}\text{C}$ were significantly more negative for short-beaked common dolphins off the Irish coast (muscle $\delta^{13}\text{C} = -17.1$, $n = 14$) compared to animals off the north coast of France (-16.5 , $n = 8$), suggesting a more oceanic/offshore diet for the former group. Similar muscle $\delta^{13}\text{C}$ values were obtained in common dolphins from Galicia, Spain (-17.0 , $n = 114$), though a slightly more negative value was obtained for common dolphins stranded in the Bay of Biscay, France (-17.4 , $n = 26$) (Chouvelon et al. 2012, Méndez-Fernandez et al. 2012). However, parameters such as age, sex, season, health and decomposition states were not assessed in these studies.

The half-life of lead (Pb) ranges from 5 to 20 years in hard tissues. As no geographic differences were found in lead concentrations or isotopic composition ($^{206}\text{Pb}/^{207}\text{Pb}$) in bone or teeth of stranded common dolphins from Brittany (NW France, $n = 17$; from a single mass stranding event [MSE] in 2002) and Galicia (NW Spain, $n = 16$), movements of animals between these continental shelf areas was proposed (Caurant et al. 2006). There were no sex-related differences in lead concentrations, though only 22% of the sample comprised males, and age was taken into account within the analysis.

Lahaye et al. (2005) proposed that there are two ecological stocks of *D. delphis* in the neritic and oceanic waters of the Bay of Biscay, based on higher renal cadmium levels in dolphins caught in the French summer tuna drift net fishery in the mid-1990s, compared to by-caught and stranded animals from French neritic waters ($n = 48$) sampled between 2001 and 2005, predominantly during

winter and spring. Cadmium has a biological half-life of more than 10 years in mammals, and higher cadmium levels in the oceanic group were attributed to the consumption of oceanic cephalopods (Lahaye et al. 2005). Cadmium tissue concentrations did not differ between males and females, in contrast to other studies (Aguilar et al. 1999). Concerns over the sample size of the 'oceanic' stock (five males, five females) proposed by Lahaye et al. (2005) were highlighted by the ICES WGMME (2009) and Murphy et al. (2009b), and they thus did not recommend a two-stock approach in the Bay of Biscay.

Murphy et al. (2007a) estimated the mean generation time for this population as 12.94 years. Consequently, taking on board recommendations by Evans & Teilmann (2009) and ICES WGMME (2012), ecological tracers showing an integration of tens of years (i.e., a few generations) should be explored. Further analysis is needed to verify the existence of ecological stocks within the Bay of Biscay, including increasing the sample size of carcasses from the proposed oceanic stock (ICES WGMME 2009, Murphy et al. 2009a). An alternative approach is the further development and application of non-lethal biomarkers from cetacean skin biopsies. Biopsy sampling from cetaceans has become a valuable approach, providing data on genetics, prey preferences, foraging ecology, contaminant loads, and physiological processes (ICES WGMME 2012, Noren & Mocklin 2012). In addition, these samples may be more representative of the population than samples collected from dead or live-stranded animals that may be ill or emaciated (ICES WGMME 2012). However, it should be noted that trace elements generally have no affinity for lipids, and few studies to date have assessed population segregation through trace element analysis of skin tissue (Kunito et al. 2002, Evans & Teilmann 2009).

Distribution and abundance

In the NE Atlantic, short-beaked common dolphins are distributed, at least during summer, from coastal waters to the mid-Atlantic ridge and from south of the Azores and the Strait of Gibraltar to around 70°N, west of Norway, but are mainly found south of 60°N (Figure 1; Murphy 2004, Cañadas et al. 2009, Murphy et al. 2009a). In the NW Atlantic, the SNESSA (Southern New England to Scotian Shelf Abundance) survey undertaken in 2007 sighted individuals as far north as 56.9°N, with higher concentrations reported on the Scotian shelf than off southern Newfoundland (Lawson et al. 2009). Short-beaked common dolphins may in fact be distributed across the whole North Atlantic Ocean, between 35°N and 55°N (partially covering a region strongly influenced by the Gulf Stream/North Atlantic Drift); however, due to a lack of observer effort west of the mid-Atlantic ridge (approximately 30–40°W), the contemporary range is unknown (Figure 1). Furthermore, the distributional boundary of the NE Atlantic population has not been determined. As outlined previously, the sampling of individuals for genetic and cranial morphometric analysis has been confined to continental shelf and slope waters and oceanic waters of the Bay of Biscay.

Cañadas et al. (2009) assessed sightings made during summer from (1) the North Atlantic Sightings Surveys (NASS), undertaken throughout most of the central and eastern North Atlantic (north of about 40°N) in 1987, 1989, 1995 and 2001; (2) the MICA93 programme (Goujon et al. 1993b); and (3) the NE Atlantic segment of the Small Cetacean Abundance in the North Sea and adjacent waters (SCANS) survey from 1994 (Hammond et al. 2002). Analyses suggested that common dolphins were most commonly sighted in water temperatures above 15°C, depths of 400–1000 m (an association with shelf features was noted), and within the area bounded by latitudes 49–55°N and longitudes 20–30°W. It should be noted that SST data were only available for some of the waters surveyed. Average group size was 15 ± 2.2 individuals (\pm standard error, SE; range 1–239) and showed a significant increasing trend with depth from 8.0 ± 1.44 animals in waters less than 400 m to 18.6 ± 2.76 animals in water depths more than 2000 m (Cañadas et al. 2009). The most northerly sighting was at 56°45' N, substantially further south than the most northerly observation for this species in the NE Atlantic at 73°34' N 11°04' E, made in August at an SST of 10.7°C (McBrearty et al.

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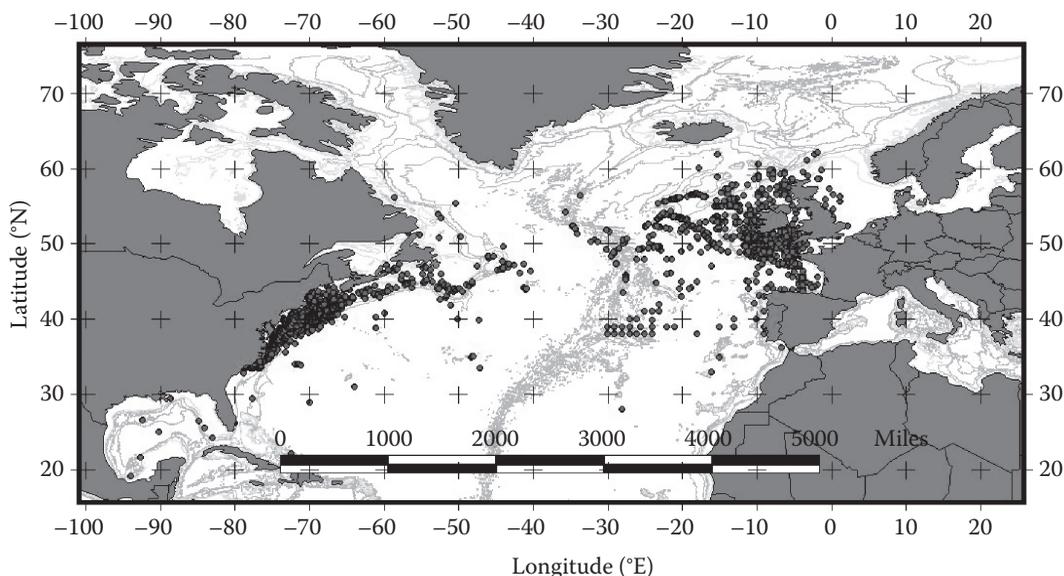


Figure 1 (See also colour figure in the insert) Distribution of common dolphin sightings in the North Atlantic (Murphy et al. 2009a). Data obtained between 1963 and 2007 by a large number of observer schemes, though the majority of sightings were obtained after 1980 and primarily during summer. Sightings data that contributed to this map were provided by Northeast Fisheries Science Center, NOAA; University of North Carolina at Wilmington; University of Rhode Island for access to BLM data; Canadian Wildlife Service for access to PIROP data; North Atlantic Marine Mammal Commission (NAMMCO), Museum of Natural History, Faroe Islands, and Marine Research Institute, Iceland, for access to NASS and T-NASS survey data; Joint Nature Conservation Committee; Sea Watch Foundation; Sea Mammal Research Unit (SMRU), St. Andrews University, for access to SCANS I data; Institut Français de Recherche pour l'Exploitation de la Mer (IFREMER) for access to MICA 1993 survey data; University College Cork for access to SIAR survey data; MAR-ECO survey data; Whale Watch Azores; and OBIS SEAMAP (Read et al. 2007).

1986, Cañadas et al. 2009). Interestingly, there was a gap in the distribution of the common dolphin in offshore waters in a rectangular area from 42°N 18°W to 48°N 12°W, waters that were surveyed in 1993 by the MICA programme (Cañadas et al. 2009).

Reid et al. (2003) mapped the distribution of common dolphins in western European waters using sightings data obtained primarily during summer, between 1978 and 1998. Highest numbers were reported in the Celtic Sea, St. George's Channel, western approaches of the English Channel, and off southern and western Ireland (Figure 2). The species was not observed in the eastern English Channel and only occasionally in the North Sea, mainly from June to September.

Further development of this data resource, including collation and inclusion of new datasets and development of statistical modelling techniques, has been initiated in recent years. Paxton & Thomas (2010) assessed common dolphin abundance and distribution in the Irish Sea between 1980 and 2009, with highest numbers occurring in St. George's Channel. They reported an increasing trend in the abundance of animals over time (Figure 3). To test the utility of such combined datasets for monitoring cetacean populations, Paxton & Thomas (2010) assessed the minimum population change that could be detected with a reasonable degree of certainty through power analysis of an index based on the ratio of population density estimates for different years (a ratio of 1 indicating no change). For common dolphin in the latest time period in the dataset (2003 to 2008), the minimum population density ratio, expressed on an annual basis, detectable with a statistical power of 0.8 was 0.978, equivalent to a 2.2% decline per year. Following from this work, the statistical techniques were further developed and data from the neighbouring Celtic Sea and the Greater Minch Area

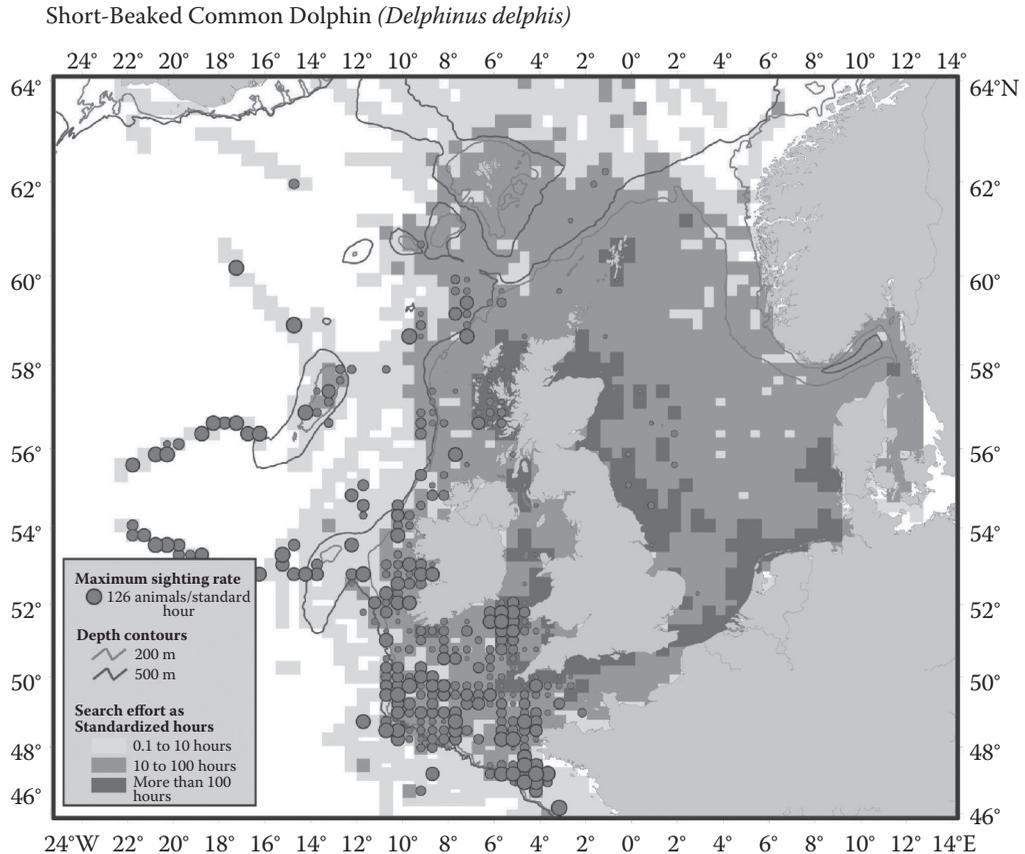


Figure 2 (See also colour figure in the insert) Distribution of common dolphins in western European waters. (Data obtained from 1978 to 1998; Reid et al. 2003. With permission.)

(western Scotland) were included (1982–2010). High population density of common dolphin was predicted throughout much of the Celtic Sea, but with high uncertainty owing to low observer effort in this region (Paxton et al. 2011). Paxton et al. (2011) noted that common dolphin abundance in the Celtic Sea, the Irish Sea, and off the west coast of Scotland generally peaked in the autumn but was also high in May and June. Numbers in the autumn peak varied between 50,200 (confidence interval [CI] = 30,800–113,600, coefficient of variation [CV] = 0.30) in 1995 and 180,900 (CI = 108,600–399,600, CV = 0.34) in 2008.

In more southern waters of the NE Atlantic, not assessed by these studies, common dolphins are one of the most frequently sighted cetaceans off the coasts of France, northern Spain, mainland Portugal and the islands of Madeira and the Azores (Silva et al. 2003, López et al. 2004, Marcos-Ipiña et al. 2005, Certain et al. 2008, 2011, Brito et al. 2009, Vieira et al. 2009, ICES WGMME 2010, Marcos et al. 2010, Pierce et al. 2010, Spyarakos et al. 2011, Moura et al. 2012). High densities in the Bay of Biscay are associated with the shelf break, though during spring common dolphins are more abundant closer to the coast, especially in areas of river plumes (Certain et al. 2008, 2011). The French PELGAS (pelagic acoustic spring) surveys between 2003 and 2008 indicated that common dolphins are distributed throughout this region in spring, with greatest abundance between the upper Gironde river plume to waters off the Vendée coast, around canyons in the south of the Bay (Cap Ferret and around), and in coastal waters off Brittany (ICES WGMME 2010, Certain et al. 2011).

THE SHORT-BEAKED COMMON DOLPHIN IN THE NORTH-EAST ATLANTIC

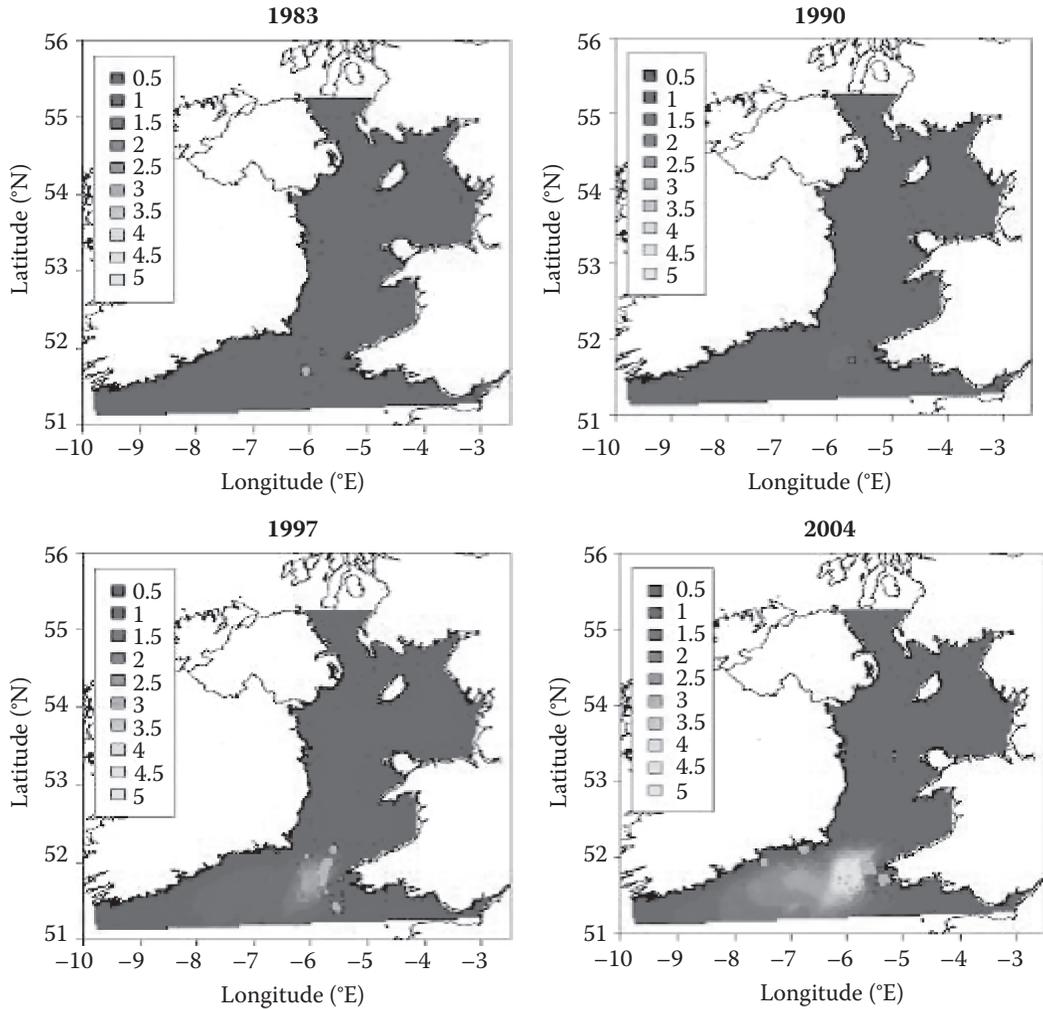


Figure 3 (See also colour figure in the insert) Predictions of common dolphin density for July 1983, 1990, 1997 and 2004 according to a two-stage modelling process. Green circles are proportional in area to estimated density of common dolphin associated with that segment locality. Numbers indicate upper bound of colour-coded densities (animals km⁻²). (From Paxton & Thomas 2010. With permission.)

In the waters off Gipuzkoan, Basque, northern Spain, common dolphins were sighted year round, predominantly in water depths of 200–1200 m, and in the vicinity of the Cap Breton canyon (Marcos et al. 2010). Densities were higher in winter and decreased significantly in spring. Data were collected from ship-based surveys carried out between April 2003 and October 2008 (Marcos et al. 2010). Off Galicia (NW Spain), short-beaked common dolphins were most commonly sighted in deeper waters (>200 m), with the highest number of sightings in the second quarter of the year (López et al. 2004). Coastal sightings of *D. delphis* were most commonly where the continental shelf was narrowest, consistent with the dolphins occupying deeper waters (Pierce et al. 2010). In surveys undertaken onboard vessels fishing on the continental shelf in 2001 and 2003 (between February and September), *D. delphis* were primarily sighted from May to August, and mean group size was 25.4 individuals (Spyrakos et al. 2011). Once survey effort was taken into account, there was no relationship between sighting frequency and water depth, and although larger group sizes

were observed in the north, off Galicia, there were fewer sightings in this area compared to the south (Spyrakos et al. 2011).

Individuals were sighted year-round off mainland Portugal, primarily along oceanic features such as canyons (Brito et al. 2009, Vieira et al. 2009). Moura et al. (2012) suggested that *D. delphis* has a patchy distribution, varying on an annual basis, associated primarily with chlorophyll concentrations, which possibly reflects a higher incidence of pelagic schooling fish in those areas. The highest encounter rates were recorded off Peniche, central Portugal (Moura et al. 2012). Off the Azores and Madeira archipelagos, the species is more frequently seen in coastal waters than off shore (Silva et al. 2003). While common dolphins are only seasonal visitors to the waters around Madeira, animals off the Azores exhibit some degree of site fidelity (Quérouil et al. 2010). In some years, there is a significant reduction in relative abundance off the Azores during summer and autumn (June to October) (ICES WGMME 2010).

Contemporary seasonal movements

Common dolphins are extremely mobile, and swimming speeds of 0.77 to 3.20 nautical miles per hour (1.43–5.93 km h⁻¹) have been estimated from radio tracking (Evans 1975, 1982), though animals can travel at 15–20 km h⁻¹ and sometimes twice as fast (Murphy et al. 2008). Maximum burst swimming speeds of 8 m s⁻¹ have been reported in captive *Delphinus delphis* (Rohr et al. 2002). Radio tracking showed that a female dolphin in the eastern Pacific Ocean travelled approximately 270 nautical miles (500 km) from the point of release over a 10-day period (Evans 1975, 1982). Two rehabilitated common dolphins were tagged with satellite-linked radio transmitters in 1994 and 1995, and one of the individuals was tracked for 31 days off the coast of California. The dolphin immediately moved off shore into deep water and began moving north. It travelled about 400 km within 5 days of release, then covered another 250 km, approximately, after which it continued travelling north until radio contact was lost (Zagzebski et al. 2006). The only other report of long-distance movement by a common dolphin was of a naturally marked individual that travelled a minimum of 1000 km across the Ionian and the Adriatic Seas (Genov et al. 2012).

In the NE Atlantic, there are no data on the habitat range of individual common dolphins, but distributional data suggest large-scale seasonal movements (ICES WGMME 2005). Analysis of sightings data collated by Reid et al. (2003) indicated that common dolphins are more widely dispersed along the continental shelf edge and in deep offshore waters, as well as off the west coast of Scotland and Ireland and in St. George's Channel, during the summer than in winter, when there are pronounced concentrations in shelf waters of the western English Channel, St. George's Channel, off shore in the Celtic Sea, and also off the coasts of NW France (western Brittany), south-western (SW) and NW Ireland and NW Scotland (ICES WGMME 2005). These seasonal movements may be related to prey availability and distribution (ICES WGMME 2005).

Brereton et al. (2005) also reported winter inshore movements of short-beaked common dolphins based on sightings data collected between 1995 and 2002 from 'platforms of opportunity', that is, ferries operating from Portsmouth, England, to Bilbao, Spain. From summer to winter, there was a 10-fold increase in the number of dolphins observed in the western English Channel. In July, highest densities and largest group sizes were seen along the northern shelf slope of the Bay of Biscay, coinciding with the mating/calving period (Murphy et al. 2005a, 2009b). Macleod et al. (2009) extended the sampling period to incorporate data collected up to 2006 and noted a similar pattern, in addition to a 5-fold increase (0.02 in 1996 to 0.11 in 2006) in wintertime occupancy in the western English Channel during the study period (though this increasing trend was not statistically significant). This increased wintertime occurrence coincided with a peak in reported strandings along the SW coast of the United Kingdom in the early 2000s (Jepson 2005) (see 'Stranding patterns' section).

Kiszka et al. (2007) also analysed sightings data collected opportunistically onboard ferries operating in the English Channel and Bay of Biscay between 1998 and 2002, predominantly from July to October. As in previous studies, summertime aggregations were larger in the northern Bay of Biscay, primarily along the shelf slope, than in the western English Channel. This may be related to the distribution of their preferred prey species in this area, *Sardina pilchardus* and *Trachurus trachurus* (Meynier 2004, Kiszka et al. 2007, Certain et al. 2011). Common dolphins also occurred seasonally off the French Channel coast, primarily around Ile d'Ouessant (western Brittany) and north of the Channel Islands, with only a few reported sightings in the eastern Channel—assessed using year-round sightings data collected between 1980 and 2000 from a variety of French sources (Kiszka et al. 2004).

Common dolphins are present in the Irish Sea at low abundance from late spring to late summer, and the distribution appears to shift southwards out of the Irish Sea during autumn and winter, though in the Celtic Deep, where higher densities exist, they remain at least until November (Evans et al. 2007, Wall et al. 2011, Baines & Evans 2012). Goold (1998) noted that the marked decrease in numbers of common dolphins off the western Wales coast between September and October suggests an autumn migration. SST distribution across the entire region was visualized using infrared satellite imagery, and it was hypothesized from these observations that the migration coincides with a break-up of the Celtic Sea Front (Goold 1998).

In the NW Atlantic, common dolphins also undertake seasonal migrations. The species is distributed from Cape Hatteras, North Carolina, north-east to Georges Bank (35–42°N) between mid-January and May, following which dolphins move on to Georges Bank and the Scotian Shelf from midsummer to autumn (Waring et al. 2011). Migrations on to the Scotian Shelf and continental shelf off Newfoundland occur when SST exceeds 11°C (Waring et al. 2011).

Long-term distribution patterns

There is evidence of changes in the distribution of the NE Atlantic population within the last century, with both an increased occurrence in more northern waters and movements into the North Sea. These shifts were observed between the 1920s and 1960s and largely since the 1990s. Increased stranding rates of common dolphins were documented in the southern North Sea during the early to mid-twentieth century along the Dutch (1920s–1960s; Bakker & Smeenk 1987) and Danish (1937–1952; including the inner Danish waters, Kinze 1995) coastlines and the eastern coast of England (1930s–1940s; Murphy et al. 2006). The increase in strandings in the North Sea partially coincided with a decline in strandings along the Irish and SW English coasts between the late 1930s and mid-1970s (Evans & Scanlan 1989, Murphy 2004, Murphy et al. 2006). This may indicate a shift in the general distribution of the species at that time (Fraser 1934, 1946, Sheldrick 1976, Evans & Scanlan 1989, Murphy 2004, Murphy et al. 2006).

Movements of common dolphins into the northern North Sea from the Atlantic occurred in the 1930s and led to an unusually large number of reported strandings along the Scottish eastern coast. Notably, this took place during an influx of the European flying squid, *Todarodes sagittatus*, a prey species of the common dolphin, into the North Sea (Fraser 1937, 1946, Evans & Scanlan 1989, Murphy et al. 2006). There was also an increase in strandings in the southern North Sea in that period, but it is unknown if this was due to some of these dolphins migrating further south or to individuals that entered the North Sea through the English Channel (Murphy et al. 2006). Based on cranial morphometric analysis, *Delphinus delphis* skulls collected in the Netherlands primarily between 1926 and 1953 were more similar to dolphin skulls collected during the last two decades from the south-west of the United Kingdom than to skulls collected in Scotland over the same time period. However, it was noted that the Scottish cranial sample may not be representative of animals inhabiting those waters in the early to mid-1900s as the distribution and abundance of the species in Scottish waters has fluctuated over the last century (MacLeod et al. 2005, Murphy et al. 2006).

There were infrequent sightings of common dolphins in the North Sea between 1978 and 1998, the majority of which were in northern British waters (Reid et al. 2003). Common dolphins were not sighted in the North Sea during either the SCANS-I or SCANS-II surveys undertaken in July 1994 and 2005, respectively. Conversely, during the 1990s and 2000s, common dolphins were documented (sightings and strandings) in both the North and Baltic Seas in Danish, German, Polish, Finnish, Swedish and Norwegian waters (ICES WGMME 2005). Six common dolphins were stranded along the Danish coastline between 2001 and 2003 and schools of up to 10 individuals were sighted (ICES WGMME 2005). Prior to this, the last reported stranding of a common dolphin in Danish waters was in 1978 (Kinze 1995), while sightings of the species were recorded in 1979, 1982, 1990 and 1996 (Kinze 1995, ICES WGMME 2005).

Øien and Hartvedt (2009) collated sightings of common dolphins in Norwegian waters recorded between 1968 and 2008. The species was sighted every year since 1976 as far north as 72°N and in almost every month apart from February and November. Sighting rates were highest between June and October, with a peak in June. There was an unusually high number of sightings (20) in 1998, the reasons for which are unknown (Øien & Hartvedt 2009). In addition to these records, common dolphins have been recorded each summer in the Moray Firth (Scottish North Sea) from 2006 to 2009, with up to 13 encounters and group sizes ranging from 2 to 450+ individuals (Robinson et al. 2010). This summertime presence has continued since 2009 (K.P. Robinson personal communication, June 2006). Macleod et al. (2005) also reported an increase in the abundance (sightings and strandings) of common dolphins off the NW Scottish coast during the period 1992 to 2003. Taken together, these data suggest that the distribution of common dolphins is once again expanding into more northern waters, including the North Sea, as apparently occurred in the early to mid-twentieth century.

Population abundance

There is no population trend information available for common dolphins in the NE Atlantic, and a lack of knowledge on the actual status of the population, based on sightings data.

Continental shelf waters

SCANS-II estimated that 56,221 (CV = 0.23) *Delphinus delphis* occupied continental shelf and slope waters of the NE Atlantic during July 2005 (Hammond et al. in press; Table 1; Figures 4 and 5). Highest densities were reported west of Ireland and Scotland, in the Celtic Sea and extending into St. George's Channel/southern Irish Sea, along the continental shelf off SW Ireland and west of Brittany, in the western English Channel, and waters off northern Spain and Portugal (ICES WGMME 2010). Earlier large-scale sightings studies, such as MICA (Goujon et al. 1993b) and SCANS-I (Hammond et al. 2002), also collected sightings data for common dolphins inhabiting oceanic and continental shelf waters (see Cañadas et al. 2009 for distribution of survey effort). However, the earlier studies are not comparable with SCANS-II since they used only a single-platform method, and they did not correct for animals missed on the survey track or for responsive movements (i.e., attraction of dolphins to the vessel).

Offshore waters

The Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA) survey, undertaken in July 2007, estimated that there were 116,709 (CV = 0.34) *Delphinus delphis* in European offshore waters (beyond the continental shelf, from off NW Scotland to NW Spain) (CODA 2009;

Table 1 Estimates of group abundance, mean group size, animal abundance and animal density (individuals km⁻²) of *Delphinus delphis* from the SCANS-II survey, July 2005

Block	Group abundance	Mean group size	Animal abundance	Animal density
B	378 (0.73)	13.0 (0.36)	4,919 (0.82)	0.040 (0.82)
N	1256 (0.58)	1.75 (0.14)	2,199 (0.60)	0.072 (0.60)
O	375 (0.69)	2.20 (0.36)	826 (0.78)	0.018 (0.78)
P	1058 (0.33)	11.6 (0.30)	15,957 (0.31)	0.081 (0.31)
Q	558 (0.98)	3.08 (0.32)	2,230 (0.87)	0.015 (0.87)
R	1266 (0.70)	9.2 (0.19)	11,661 (0.73)	0.302 (0.73)
W	1470 (0.29)	12.3 (0.27)	18,039 (0.23)	0.130 (0.23)
Z	314 (0.84)	1.25 (0.20)	392 (0.86)	0.012 (0.86)
Total	6675 (0.27)		56,221 (0.234)	
	[3969–11,230]		[35,748–88,419]	

Source: Hammond et al. in press.

Note: Coefficients of variation are given in parentheses. Figures in square brackets are 95% confidence intervals. There were no sightings of *Delphinus delphis* in blocks H, J, L, M, S, T, U, V and Y. See Figure 4 for SCANS-II survey map.

Table 2, Figure 4). Highest densities were observed in more southern areas of the surveyed region, with most sightings along the continental slope off western France and northern Spain (Figure 6).

The small numbers sighted off the western coast of Ireland during CODA were comparable to an earlier survey undertaken in July 2000; see Figure 4 for realized survey effort during the CODA survey and note low survey effort off the SW coast of Ireland. SIAR (Survey in Western Irish Waters and the Rockall Trough), which surveyed waters over the shelf break to the north and west of Ireland, estimated only 4496 individuals in this region (Ó Cadhla et al. 2003); see ICES WGMME (2005) for a map of summer abundance surveys in the NE Atlantic. The study area was about 120,000 km², covering the western Irish continental shelf, central and eastern Rockall Trough and from the Porcupine Bank to the Outer Hebrides. Results from both SIAR and CODA are in contrast to the large numbers of short-beaked common dolphins sighted off the west of Ireland during the 1990s. Faroese scientists who participated in the NASS 1995 survey covered two large areas (NASS-east and NASS-west) to the west of Ireland and Scotland. The estimated abundance of *D. delphis* in NASS-west, an area that extended beyond the CODA survey region, was 273,159 (CV = 0.26; 95% CI = 153,392–435,104) (this estimate was corrected for animals missed on the survey track and for responsive movement) (Cañadas et al. 2009). An abundance of 77,547 *D. delphis* was estimated for NASS-east, but due to limitations of the survey, this estimate was not considered reliable (Cañadas et al. 2009). Nevertheless, even allowing for the uncertainty in this estimate, it suggests a considerably greater number may have been present than during the SIAR survey.

The Trans North Atlantic Sightings Survey (T-NASS) was carried out at the same time as CODA but surveyed waters further to the north and off shore (Lawson et al. 2009). Few short-beaked common dolphins were sighted in areas where animals were seen in high abundance during the NASS 1995 survey. In 2009, the IWC Sub-Committee on Small Cetaceans identified several potential reasons for the observed changes in density/distribution, including (1) differences in sighting conditions (e.g., sea state), (2) uncertain species identification (as other dolphin species were sighted), (3) a true reduction in common dolphin density, (4) ship effect and (5) interannual distributional shifts. In addition, due to poor weather conditions during T-NASS, some of the planned survey tracks were not covered (IWC 2009).

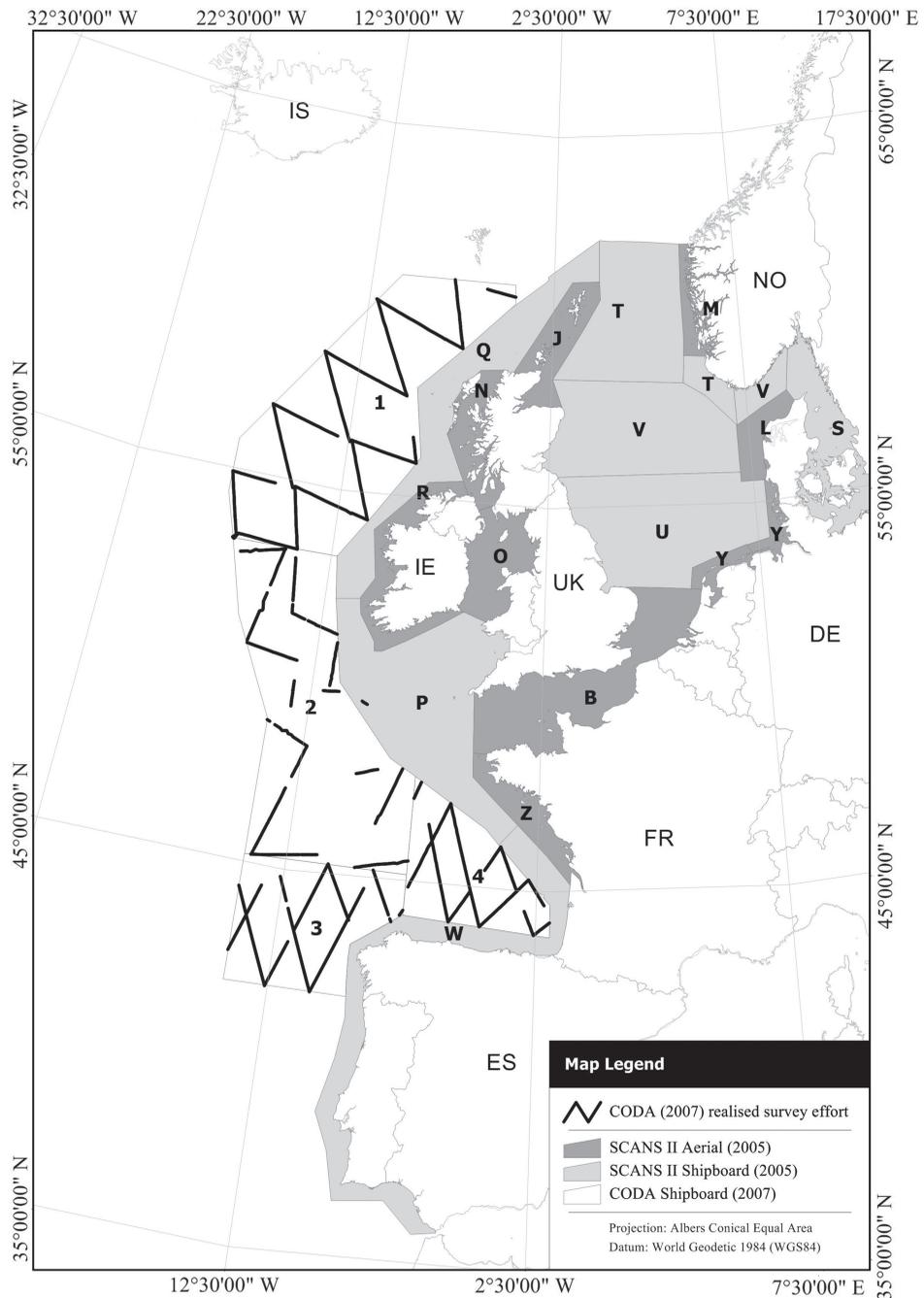


Figure 4 Survey blocks defined for the SCANS-II (undertaken in 2005) and CODA (undertaken in 2007) surveys. SCANS-II Blocks S, T, V, U, Q, P and W were surveyed by ship. The remaining blocks were surveyed from aircraft (SCANS-II 2008). CODA survey region divided into the survey blocks 1–4 and realized survey effort route (in black) (CODA 2009). (Map produced by Rene Swift from a projection of Albers conical equal area and datum from World Geodetic System 1984.)

THE SHORT-BEAKED COMMON DOLPHIN IN THE NORTH-EAST ATLANTIC

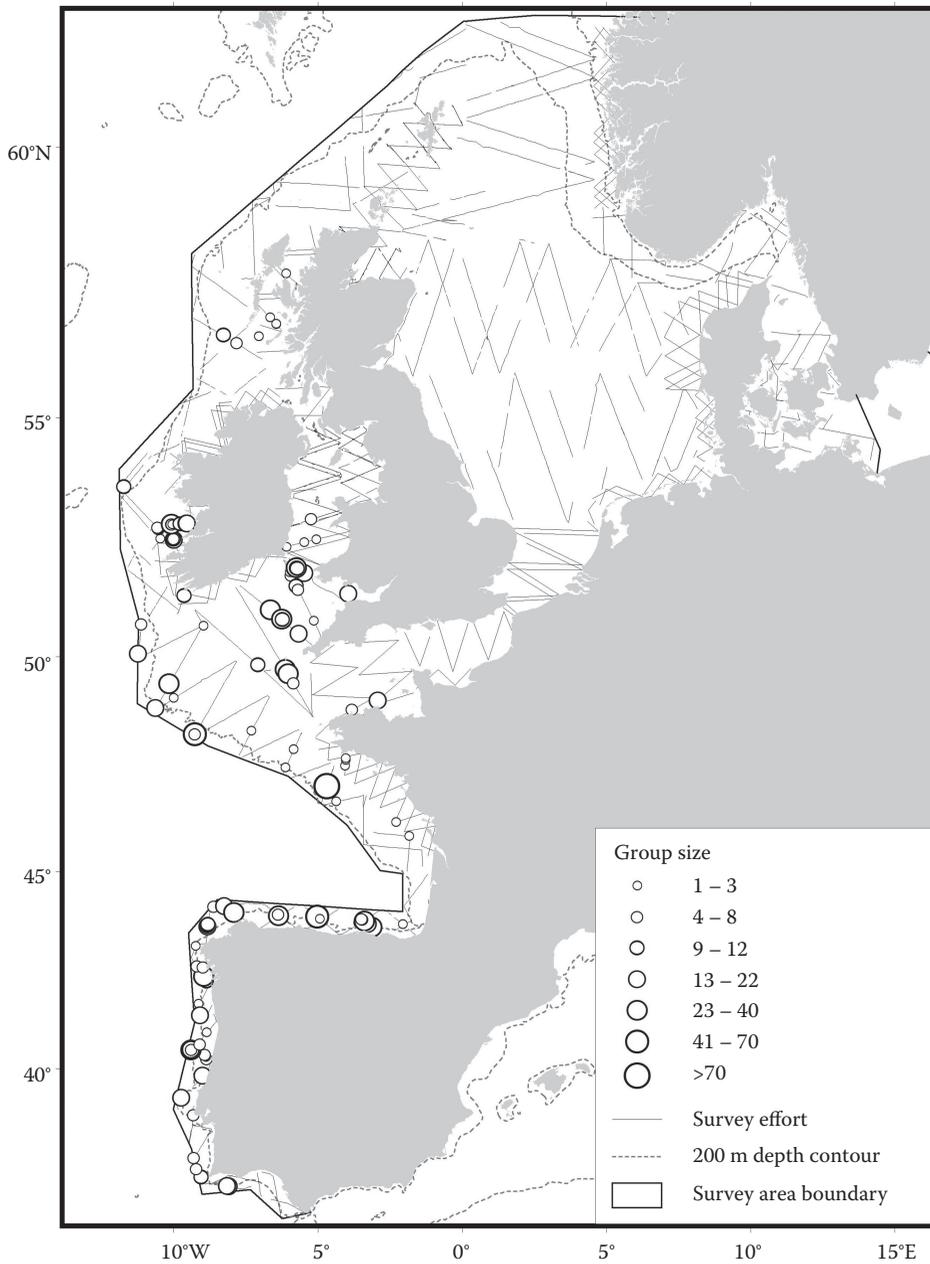


Figure 5 Sightings of common dolphins during SCANS-II in July 2005. (From Small Cetaceans in the European Atlantic and North Sea (SCANS-II), Final report to the European Commission under contract LIFE04NAT/GB/000245. St. Andrews, UK: Sea Mammal Research Unit, 2008; Hammond et al. in press, with permission from Elsevier.)

Table 2 Estimates of model-based (density surface modelling) animal abundance estimates, with coefficients of variation (CV) in brackets, and 95% confidence intervals, of *Delphinus delphis* from the CODA survey, July 2007

Block	Animal abundance (CV)	95% Confidence interval
1	4,216 (0.57)	1,478–12,027
2	52,749 (0.39)	25,054–111,059
3	21,071 (0.51)	8,270–53,689
4	38,673 (0.46)	16,464–90,839
Total	116,709 (0.34)	61,397–221,849

Sources: Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA), 2009; ICES WGMME (2010).

Note: See Figure 4 for CODA survey blocks.

Life-history parameters

Extensive studies have been undertaken since 2000 assessing life-history parameters in the NE Atlantic *Delphinus delphis* population using samples from stranded and by-caught dolphins, funded by both national governments and the European Union's Fifth and Sixth Framework Programmes for Research, including large-scale projects such as BIOCET (Bioaccumulation of Persistent Organic Pollutants in Small Cetaceans in European Waters: Transport Pathways and Impact on Reproduction) and NECESSITY (Nephrops and Cetacean Species Selection Information and Technology). These have resulted in preliminary data on a large number of biological parameters, such as the population pregnancy rate and average age at sexual maturity, essential for effective conservation management. However, these could not be considered baseline data as there may have been anthropogenic impacts prior to these studies.

Size and morphology

In the early to mid-1900s, maximum body lengths of 250 to 270 cm were recorded for short-beaked common dolphins in the NE Atlantic (Harmer 1927, Fraser 1934, 1946, 1974). If these earlier studies identified the species correctly, there may be indications that maximum length is declining within the region. During the last 30 years, maximum lengths of 250 and 239 cm have been reported for males and females, respectively, though the majority of individuals were less than 230 cm (Collet 1981, Silva & Sequeira 2003, Murphy 2004, Murphy et al. 2006, 2009b, Murphy & Rogan 2006). Average length of newborn calves is 93 cm (range 89–110 cm) (Collet 1981, Murphy & Rogan 2006).

Short-beaked common dolphins in the NE Atlantic exhibit sexual size dimorphism, with males significantly larger than females in total body length and 19 of 23 other morphometric characters (Murphy & Rogan 2006). Although there is a statistically significant difference in size, it is only moderate: Murphy & Rogan (2006) calculated a sexual size dimorphism ratio of 1.06 using average adult body lengths of 201.2 cm for females and 212.9 cm for males. Sexual shape dimorphism (relative size) was not detected in body characters, apart from the presence of prominent postanal humps in mature males (Murphy & Rogan 2006). Interestingly, unlike in spinner (*Stenella longirostris*) and spotted dolphins (*S. attenuata*), the postanal hump in the common dolphin is composed of muscle and not connective tissue, suggesting different functions (Murphy et al. 2005a). It has been proposed that the postanal hump in the common dolphin may serve in female choice, allowing the identification of the healthiest male, that is, the male that can produce the largest quantity of sperm (Lewis 1991, Neumann et al. 2002, Murphy et al. 2005a) since preliminary investigations suggested that the size of the postanal hump is positively correlated with testis size (Lewis 1991).

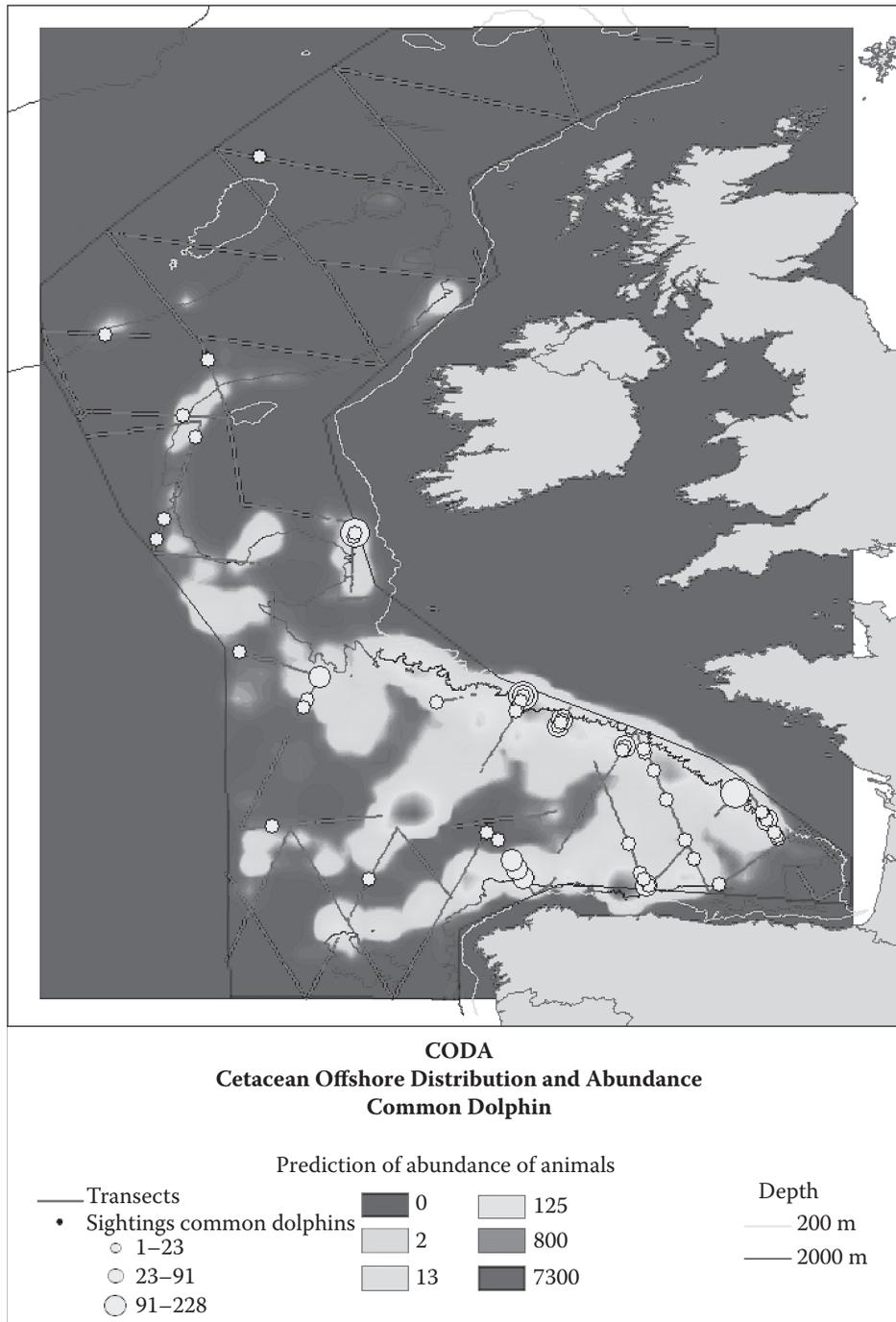


Figure 6 (See also colour figure in the insert) Surface maps of smoothed predicted abundance of common dolphins in offshore waters, including the distribution of sightings (circles proportional to group size). (Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA) 2009.)

Male short-beaked common dolphins are significantly larger than females in condylobasal length (CBL) and 21 other cranial morphometric characters (Murphy 2006). Sexual dimorphism in cranial shape was identified in seven characters, mainly related to the width of the skull. Mature males had (in relation to CBL) significantly greater rostrum width at base, zygomatic width, post-orbital width, width between temporal fossae, maximum width of external nares, orbital length, and mandible depth.

Population parameters

Average age at the attainment of sexual maturity was estimated at 11.9 years in males, based on examination of common dolphins sampled by the Irish and French stranding and by-catch observer programmes between 1991 and 2003 (Murphy et al. 2005a). Sexually mature males ranged from 195 to 233 cm in length and 8 to 28 years in age. Applying a single Gompertz growth curve to the male age data produced an asymptotic length of 206 cm (Murphy et al. 2005a). Murphy et al. (2009b) analysed a much larger sample size of stranded and by-caught female common dolphins collected throughout the NE Atlantic (Scotland to Portugal) between 1990 and 2006. Female body lengths ranged from 91 to 239 cm, and the maximum estimated age was 30 years (Murphy et al. 2009b, 2010). The asymptotic length, estimated using Richard's model, was 202 cm (Murphy et al. 2009b). Average age and length at sexual maturity were 8.2 years and 188 cm, respectively (Murphy et al. 2009b).

Based on mortality data from 248 mature females, an annual pregnancy rate of 26% and extended calving interval (gestation, lactation and resting periods) of 3.8 years were determined for the NE Atlantic population (Murphy et al. 2009b). There was no significant difference in the proportion of pregnant females between different geographical areas (Ireland, United Kingdom, France, NW Spain) of the NE Atlantic. The pregnancy rate was also estimated using a control group of 'healthy' individuals, that is, individuals not suffering from any infectious or non-infectious disease that may inhibit reproduction. As no significant difference was found in proportion of pregnant females between the control group and the whole sample, it appears that the sampling of stranded and by-caught short-beaked common dolphins is adequate for estimating population reproductive parameters. This is in contrast with other cetacean species, such as the harbour porpoise, with a large number of stranded individuals dying due to ill health or poor condition, including starvation, disease, and bacterial and parasitic infections (Deaville & Jepson 2011). In contrast, the majority of stranded common dolphins in this region were killed in fishing gear (by-catch) and subsequently washed ashore (Murphy et al. 2009b, Deaville & Jepson 2011).

The reproductive rate identified in the NE Atlantic population is relatively low compared to other *Delphinus* sp. populations, in which pregnancy rates are higher, such as South Africa, 40.2% (Mendolia 1989, Murphy et al. 2009b); or ETP, 47% (Danil & Chivers 2007). Murphy et al. (2009b) found no evidence of compensatory density-dependent responses in reproductive parameters. No significant differences were observed in the proportion of pregnant females, proportion of mature females simultaneously pregnant and lactating, average age attained at sexual maturity, or nutritional condition of females between two different time periods (1991–1999 and 2000–2006).

In the NE Atlantic, short-beaked common dolphins exhibit reproductive seasonality. A unimodal calving/mating period extends from April to September, possibly with a more active period in July and August (Murphy et al. 2005a, 2009b). Estimated individual conception dates of sampled fetuses ranged from 5 April to 2 October, though the average date of conception was 19 July, and 40% of individuals were conceived during this month (Murphy et al. 2009b). The gestation period in the population was estimated at 0.99 years (Murphy et al. 2009b). Even though the sample size was small, sexually mature and pubertal females were reported ovulating only during May to September (6 of 45 individuals examined) (Murphy 2004). Such an extended mating period in the NE Atlantic population allows females to undergo numerous ovulations, with some individuals possibly completing up to five reproductive cycles during this period (Murphy et al. 2010). This

provides a substantial buffer for individuals that may not conceive during their first oestrous within the mating period.

Male gonadal tissue in this region also exhibits seasonality, evidenced by reduced testis weights and testicular cellular activity outside the mating period (Murphy et al. 2005a, Murphy & Rogan 2006). In the NE Atlantic, mature male common dolphins developed large testes, relative to their body size, with combined testes weight ranging from 415.9 to 5000 g. Macleod (2010) ranked common dolphins comparatively high among 31 cetacean species in their relative investment in testicular tissue, based on an assessment of both percentage testes (3.2% of body mass in the species) and residual testes mass (deviation from the mean testes mass-body mass relationship of 31 cetacean species). The presence of enlarged testes and the existence of moderate sexual dimorphism in the species suggest postmating competition among males (i.e., sperm competition), resulting from a promiscuous mating system (Murphy et al. 2005a). To date, there have been no studies of mating strategies in wild common dolphins in this region to verify this hypothesis. Males continue to produce sperm outside the mating period (Murphy et al. 2005a), which possibly enables them to partake in 'recreational' (non-reproductive) mating. Recreational mating has been noted in New Zealand waters (Neumann 2001, Murphy et al. 2005a), but the function of this behaviour is not known.

Age and sex segregation

Common dolphins are found in a wide range of group sizes, up to 1000 to 5000 individuals (Murphy 2004 and references therein). There is evidence that smaller groups are segregated by age and sex, especially during winter (i.e., outside the mating period). Three mass live strandings (three or more individuals) of 'nursery groups' have been reported along the Irish, French, and UK coastlines. In February 2001, there were 15 common dolphins stranded alive on the Mullet Peninsula, western coast of Ireland. Eleven dolphins were refloated, and five died, including one male yearling, three sexually mature females aged between 14 and 17 years and one pregnant 17-year-old female (Murphy 2004). Further evidence of nursery groups within this region arose when a mass live-stranding event involving approximately 100 individuals took place at Pleubian, France, in 2002. The animals that died comprised one male calf and 52 females aged between 0.5 to 2.8 years and 6 to 26 years (Dabin et al. 2008, Viricel et al. 2008). This suggests that weaned juveniles, subadult males and mature males segregate, at times, from nursery groups. Interestingly, genetic analysis of this nursery group revealed that variability within the mass-stranded group was similar to variability observed in single strandings of common dolphins along the French coastline; that is, mature females within the nursery group were not necessarily genetically related (Viricel et al. 2008). In June 2008, 26 *Delphinus delphis* died during a mass stranding in Cornwall, SW England (Jepson & Deaville 2009). The group comprised five lactating females and sexually immature individuals of both sexes, ranging in age from 2 to 9 years (Jepson et al. in press). Finally, a group of seven common dolphins live stranded in May 2002 on the western coast of Ireland resulted in the deaths of four individuals (two males and two females), ranging in age from 1 to 8.5 years (Glanville et al. 2003, Murphy 2004), which provides further evidence that juveniles and subadults may segregate, at times, from other social groups.

Examination of individuals incidentally caught during spring and summer in Portuguese gill nets, beach seine nets and trawls revealed that sexually mature females only associated with young calves, and sexually immature males either formed separate groups (sometimes with small numbers of immature females) or joined mature male groups (Silva & Sequeira 2003). There was almost a complete absence of sexually immature females in the by-catch (Silva & Sequeira 2003). In contrast, an assessment of dolphin by-catch in the Irish and French albacore tuna (*Thunnus alalunga*) drift net fishery suggested mixing of nursery and mature male bachelor groups, though this was during the mating period (Murphy & Rogan 2006). The NE Atlantic albacore tuna drift-net fishery

operated over a large area, primarily beyond the continental shelf (see ‘Fisheries interactions’ section). The lack of juveniles and subadults (3 to 8 years) incidentally caught in this type of gear, which has low species selectivity above a minimum body size (Northridge 2009), suggests a significant summertime segregation of juveniles and subadults from other groups over a large geographic area, though this requires further investigation.

Mirimin et al. (2012) assessed group composition of *Delphinus delphis* that mass-stranded on the western coast of Ireland or were by-caught in the Irish albacore tuna and UK bass (*Dicentrarchus labrax*) fisheries; these authors used 14 microsatellite loci and 360 bp of the mtDNA control region. Parentage and kinship analyses revealed that dolphins caught in the same net tended to be unrelated to each other, with the exception of mother-offspring pairs (Mirimin et al. 2012). Individuals from the same group rarely shared the same mtDNA haplotype, apart from mother-offspring pairs, indicating the presence of multiple maternal lineages.

Feeding ecology

In the wild, several distinct feeding strategies have been described for individual common dolphins, including high-speed pursuits, ‘fish whacking’ (striking with the tail) and ‘kerplunking’ (rapid tail movement on the surface) (Neumann & Orams 2003, Burgess 2006), while cooperative feeding allows dolphins to exploit shoaling prey in an energetically advantageous way (Young & Cockcroft 1994, Brophy 2003). Coordinated feeding strategies includes ‘carouselling’, line abreast, wall formation, synchronous diving and bubble cloud production (Neumann & Orams 2003, Peschak 2005, Burgess 2006). Results from studies of feeding behaviour in New Zealand suggest that variation in prey distribution and productivity, possibly as a result of differing habitats (i.e., shallow waters vs. open ocean), may affect strategy selection (Neumann & Orams 2003, Burgess 2006, Stockin & Orams 2009).

Common dolphins in the NE Atlantic have been observed in mixed feeding aggregations comprising other cetaceans (e.g., *Stenella frontalis*, and *Tursiops truncatus*), large tunas and seabirds (Clua & Grosvalet 2001). In this region, common dolphins have shown associations with albacore tuna, though it is not known how long these associations last. The stomach contents of *Delphinus delphis* caught in drift nets set for albacore tuna included all the prey species (fish) found in the stomachs of tuna (Hassani et al. 1997). In the ETP, the strong associations between yellowfin tuna *Thunnus albacares* and pantropical spotted dolphins *Stenella attenuata* were attributed to the risk of predation, resulting in these species forming large, mixed-species groups, and not due to feeding advantages (Scott et al. 2012). At dawn and dusk off the Azores, feeding aggregations initiated and sustained by common dolphins feeding at the periphery of bait balls and actively herding fish towards the surface can be broken down by the arrival of large tuna (*Thunnus thynnus*, *T. albacares*) as they swim straight into the bait ball, foraging on fish (Clua & Grosvalet 2001). This suggests that the tunas benefit from these aggregations by accessing prey more easily. Most of the information on diet of common dolphins in the NE Atlantic arises from studies of stomach contents of stranded and by-caught individuals. While these studies have been highly informative, they are limited by sampling biases. They provide information on dietary preferences of individuals inhabiting inshore waters primarily during winter, when most strandings occur, and of by-caught dolphins that were either feeding on target prey species of a particular fishery or opportunistically exploiting enhanced prey availability around fishing operations, including non-target species of those fisheries. For example, common dolphins have been reported to feed both on discards and directly from the cod end of trawls and inside trawl nets on small non-target prey species (see ‘Fisheries interactions’ section). In addition to seasonal and age-sex biases in sampling, differences in the gut passage times and rates of digestion of different prey species may create additional limitations and biases (Pusineri et al. 2007). Recently, fatty acid and stable isotope signatures have been used to discriminate spatial and temporal differences in diet in the NE Atlantic population of *Delphinus delphis*.

Analysis of non-lethal biopsies of tissues such as blubber and muscle may provide a good general overview of the diet of this species. Variables such as age, sex and health status should be taken into account when interpreting results from these studies (see ‘Ecological stocks’ section).

Temporal, geographic and seasonal variations in diet

Common dolphins are opportunistic feeders (Young & Cockcroft 1994), though more recently it has been suggested that they select prey based on energy densities (Santos et al. 2004, Brophy et al. 2009, Spitz et al. 2010). In the NE Atlantic, the diet of common dolphins includes a wide variety of fish and squid species, though it is predominantly composed of a few main species that vary with season and region (Murphy et al. 2008). In areas where preferred prey species are in high abundance, common dolphins tend to select those species. Consequently, diet displays strong interannual and seasonal variation (Murphy et al. 2008). During winter, common dolphins in inshore waters prey mainly on shoaling pelagic fishes, whereas in summer, *Delphinus delphis* caught in tuna drift nets set at night beyond the continental shelf edge had fed predominantly on squid and mesopelagic fishes, such as lanternfish (Myctophidae). Other small delphinids in the NE Atlantic show similar diet plasticity. For example, striped dolphins, *Stenella coeruleoalba*, switch between migrating mesopelagic prey to neritic or coastal prey types (Spitz et al. 2006). It is not known whether common dolphins in the NE Atlantic follow the migratory patterns of their preferred prey, as dolphins have not been fitted with animal-borne tracking devices in this region. However, inshore movements of common dolphins into the Celtic Sea and western English Channel in winter have been attributed to feeding opportunities (see ‘Distribution and abundance’ section).

Offshore waters

Common dolphins caught in Irish and French albacore tuna drift nets during the 1990s (see ‘Fisheries interactions’ section) were predominantly feeding at night, when the migrating deep scattering layer approaches the surface (Hassani et al. 1997, Pusineri et al. 2007, Brophy et al. 2009). The dietary preferences of these by-caught common dolphins were assessed by two different studies. Pusineri et al. (2007) examined stomach contents from 63 *Delphinus delphis* caught in the French tuna drift nets from 39°N to 50°N and 10°W to 21°W. Animals sampled were biased towards younger dolphins, and age-sex differences in dietary preferences were not assessed in the study. Brophy et al. (2009) analysed dietary remains from 58 *D. delphis* caught in Irish drift nets, predominantly along and off the continental slope, SW of Ireland. The main prey species of animals caught in French drift nets were the myctophids *Notoscopelus kroyeri*, *Benthosema glaciale*, and *Myctophum punctatum*; the sternoptychid *Maurollicus muelleri*; and cephalopods *Ancistroteuthis lichtensteinii*, *Gonatus steenstrupi*, *Brachioteuthis riisei*, and *Teuthowenia megalops* (Pusineri et al. 2007). A number of these are small schooling species, a similar prey profile to that consumed in inshore waters of the Bay of Biscay (Pusineri et al. 2007). *Notoscopelus kroyeri* dominated the diet, occurring in 84% of stomachs, and accounted for 65% of abundance by number. Similar prey types were observed in the stomachs of *Delphinus delphis* caught in Irish drift nets. Myctophids were the most common family of fish and comprised *Myctophum punctatum* (29% of all prey items) and *Notoscopelus kroyeri* (22%). Of lesser importance was the main cephalopod species consumed, *Brachioteuthis riisei*. Other species included *Arctozenus risso*, *Maurollicus muelleri*, *Benthosema glaciale* and *Gonatus steenstrupi*. Interestingly, the horse mackerel (*Trachurus trachurus*, Carangidae) was the numerically dominant species (38%), with a high index of importance, though was listed only fourth in order of significance expressed as percentage frequency of occurrence. A few prey species were common to the stomachs of dolphins caught off shore and animals stranded along the Irish coast, a situation not observed in the Bay of Biscay (Meynier 2004, Pusineri et al. 2007). These prey species included horse mackerel, blue whiting (*Micromesistius poutassou*), *Benthosema glaciale* and *Brachioteuthis riisei* (Brophy et al. 2009).

Along the Mid-Atlantic Ridge, Doksaeter et al. (2008) assessed spatial correlations between dolphin occurrence and candidate prey organisms that were recorded acoustically and sampled by midwater trawling. This indicated that mesopelagic fishes such as *Lampanyctus macdonaldi*, *Stomias boa ferox* and *Chauliodus sloani* were important prey for common dolphins.

Inshore waters

Ireland The diet of 76 stranded common dolphins (including by-caught dolphins that subsequently stranded; 21%) and individuals retrieved directly from fishing gear (9%), sampled over a 15-year period (1990–2004), was primarily composed of fish (97% by number, cephalopods 3% by number) represented by at least nine families and 14 species (Brophy et al. 2009). Gadoids were the best-represented family, with *Trisopterus* spp. (*T. esmarkii* and *T. minutus*) the most important species. *Trisopterus* spp. were the most common prey during the summer (April to September) and winter (October to March). Although gobies appeared to dominate the diet during the winter, this was largely due to the occurrence of 1822 otoliths in one stomach. Other fish species of importance were blue whiting, whiting (*Merlangius merlangus*), *Argentina* sp., Atlantic herring (*Clupea harengus*), and the European sprat (*Sprattus sprattus*). The average prey size present in stranded common dolphins (9.7 cm, SD = 6.45 cm) was considerably larger than that found in the stomachs of dolphins caught in Irish tuna drift nets (4.2 cm, SD = 2.25 cm). For whiting, 40% of the fish present in the diet were above the commercial minimum landing size (Brophy et al. 2009).

United Kingdom Common dolphins off the SW coast of the United Kingdom consume a wide variety of fish, but primarily sardine *Sardina pilchardus*,* mackerel, *Scomber scombrus*, horse mackerel, Norway pout, *Trisopterus esmarkii*, other clupeids and various squid species (Pascoe 1986, Kuiken et al. 1994, Natural History Museum 1995, Gosselin 2001). Gosselin (2001) assessed non-empty stomachs from 18 by-caught dolphins that were stranded along the SW coast of the United Kingdom between December 2000 and April 2001. Sardine and horse mackerel comprised 40% and 37% of the stomach contents, respectively, with mackerel and Norway pout found to a lesser extent. Diets were similar to those of by-caught common dolphins that were mass stranded along the SW coast of the United Kingdom during the first quarter of 1992 (Kuiken et al. 1994). Mackerel and pilchards dominated the diet; the size of the latter was remarkably large, ranging from 14 to 30 cm in length. In Scottish waters, 14 fish taxa and 2 cephalopod taxa (*Alloteuthis subulata*, unidentified Sepiolidae) were identified in the stomachs of nine common dolphins that were stranded between 2000 and 2003. Mackerel, followed by whiting, were the main prey consumed, together making up more than 40% of the estimated prey weight (Learmonth et al. 2004b).

France Stomach contents analysis of 26 common dolphins that were stranded along the Normandy coast (English Channel) revealed they consumed mainly gadoid fish (*Trisopterus luscus*), gobies and mackerel (De Pierrepont et al. 2005). Cephalopods occurred in small numbers in the diet. In inshore waters of the Bay of Biscay, four taxa contributed to the majority of the dietary remains of 71 common dolphins that were stranded between 1999 and 2002. These included sardine, anchovy *Engraulis encrasicolus*, sprat and horse mackerel, which represented 44.9%, 22.6%, 8.0% and 5.0% by mass of the fresh diet, respectively (Meynier et al. 2008). Cephalopods only constituted 5.6% of the diet by mass, comprising *Alloteuthis* spp., *Loligo* spp., and unidentified Loliginidae. Meynier et al. (2008) reported that the diet displayed strong seasonal and interannual variations in terms of both prey species composition and prey size distributions, reflecting prey availability in the area. However, estimated daily food intakes changed relatively little, as all diets included a high

* The UK Sea Fish Industry Authority classifies sardines as young pilchards. One criterion suggests fish shorter in length than 15 cm are sardines, and larger ones are pilchards.

proportion of lipid-rich fish (sardine, sprat, anchovy and mackerel; 73 to 93% by mass). Sardines were the dominant prey by mass in summer/autumn and winter, whereas the quantity of sprats was highest in spring. Horse mackerel was absent from the diet during the summer, and anchovy did not show any marked seasonal variation, though female common dolphins did consume more of this prey type during the summer. As noted in Irish waters, gobies were the most important prey by number during the winter (to summer), though it is not known if these small demersal fishes were primary prey or prey of other fish secondarily eaten by the dolphins (Meynier et al. 2008).

Spain Off Galicia, blue whiting and sardine together comprised 56.5% of reconstructed prey weight present in the stomachs of 414 stranded individuals (one-third of which showed signs of net entanglement), sampled predominantly during winter from 1991 to 2003 (Santos et al. 2004). Gobies were the most numerous prey group. Other species included horse mackerel, sand smelt *Atherina* sp., and cephalopods such as *Loligo* sp. In total, 25 fish species and 15 cephalopod species were identified in the dietary remains. Although the diet was primarily composed of only two species, there was evidence that common dolphins off Galicia were opportunistic feeders (Santos et al. 2004). For example, higher numbers of sardines were consumed in years of higher sardine abundance and lower recruitment of blue whiting. This was possibly due to sardines having a higher calorific value than blue whiting (Santos et al. 2004). Strong seasonal variations were also observed in dietary preferences, with higher numbers of gobies, *Atherina* sp. and small squid (*Alloteuthis* sp.) consumed during the first quarter. On the whole, dolphins were exploiting different size classes of prey compared to those targeted by fisheries, apart from sardines (15.5- to 22.5-cm long), which were well above the minimum landing size of 11 cm (Santos et al. 2004). A recent stable isotope study has confirmed the importance of blue whiting and sardines in the diet of common dolphins off Galicia (Méndez-Fernández et al. 2012).

Portugal Silva (1999) analysed stomach contents from 50 stranded and by-caught common dolphins sampled between 1987 and 1997. Even though 27 different fish species and 8 cephalopod species were identified, the diet was mainly composed of a small number of taxa (Silva 1999). Six fish species (sardine, blue whiting, *Atherina* sp., horse mackerel and scombrid species) composed 84% of the total estimated weight, and sardines were the most important prey item, occurring in 81% of stomachs (27% of prey by number and 43% of estimated weight). Common dolphins stranded along the Portuguese coast appear to have a higher proportion of sardines in their diet than animals stranded along the Galician coastline (Santos et al. 2004). Spring acoustic surveys carried out by both countries since 1986 showed that sardines are more common in Portuguese waters than off Galicia (Carrera & Porteiro 2003, Santos et al. 2004).

Age- and sex-related dietary requirements

Dietary studies indicate that weaning can commence between 3 and 6 months after birth (Brophy et al. 2009). Studies of reproductive parameters, however, suggest that females may lactate for up to 10 months after parturition (Murphy 2004), and the length of the lactation period may increase with maternal age, as noted in other delphinid populations, including common dolphins elsewhere (e.g., Danil & Chivers 2007).

Few studies have assessed differences in dietary preferences between age-sex maturity classes. Meynier et al. (2008) reported that there were significant differences between age-sex maturity groups of common dolphins stranded in the Bay of Biscay, in terms of both prey species composition and prey size distributions. Prey length was weakly correlated to dolphin length, and prey composition of mature males was less diverse than that of mature females and immature dolphins, with mature males predominantly consuming sardines. Adult males also fed on larger sardines than the other age-sex groups. Off the Portuguese coast, limited variation in dietary preferences was

observed between different age-sex maturity groups. Sardine was the dominant prey species in the diet of all groups, with the exception of immature males, which ate more blue whiting (Silva 1999). Interestingly, both Silva (1999) and Meynier et al. (2008) reported that cephalopods were a minor component of the diet of mature males. Diets of *Delphinus delphis* in both offshore (summer) and inshore (predominantly winter) habitats in Irish waters revealed no significant difference in total prey numbers, prey species number, or proportion of cephalopods in the diet between different age-sex maturity groups, except that the stomachs of stranded females contained a significantly higher number of prey items than stranded males (Brophy et al. 2009). In addition, there were positive relationships between dolphin body length and both total prey numbers and number of prey species in the offshore group.

The difference in diets between inshore and offshore areas recorded by Brophy et al. (2009) provides evidence of seasonal offshore-onshore movements of common dolphins in the NE Atlantic. The energy requirement of (pregnant and) lactating *D. delphis* and their calves may contribute to the offshore movement of some mature individuals (and calves) during the spring and summer to take advantage of nutrient-rich prey at times when neritic prey are nutrient poor (or have dispersed to/from spawning grounds) (Brophy et al. 2009). Many species of pelagic fish, such as *Trisopterus* spp., whiting, sprat, and Atlantic horse mackerel, spawn to the south and SW of Ireland during the spring and summer and then migrate north (Jákupsstovu 2002, Dransfeld et al. 2004, Brophy et al. 2009). During spawning, the lipid content of these fish species falls, reducing their calorific value as prey (Brophy et al. 2009). Interestingly, the offshore dietary sample of horse mackerel comprised fish less than 1 year of age (i.e., prespawning stage). Myctophids, which are consumed in offshore waters during the summer, are reported to have higher lipid content than other marine fish species (Saito & Murata 1998, Lea et al. 2002, Brophy et al. 2009), though the size range (age class) of the myctophids that are consumed is unknown. As noted previously, the general absence of juvenile/subadult *Delphinus delphis* in the by-catch of Irish tuna drift nets suggests that they were not present in the area where this fishery operated (Murphy & Rogan 2006), which may suggest a different feeding strategy in summer. Off Portugal, immature males were found to consume blue whiting and showed a tendency to be caught in pelagic trawls targeting that species (Fernández-Contreras et al. 2010) (see 'Fisheries interactions' section).

A subsequent study assessing energy requirements of common dolphins in the Bay of Biscay has confirmed the selection of high-quality foods during summer. As noted, the diet of individuals sampled during the 1990s in this region was dominated by the myctophid *Notoscopelus kroyeri*, a high-energy prey species (Spitz et al. 2010). Surveys of the epi- to mesopelagic oceanic fish community off the Bay of Biscay in October 2002, 2003 and 2008 revealed that the alepocephalid *Xenodermichtys copei*, a low-energy prey that was not consumed by common dolphins, was the most abundant species (Spitz et al. 2010).

Health status and cause of death

National marine mammal stranding networks have for many years collected basic data (date, location, species, etc.) on single and mass stranding events in many countries. Some of these datasets are now nearly 100 years old. However, it is only in recent years that more-detailed investigations have been conducted on stranded animals through systematic necropsies of stranded carcasses. These new studies have provided new insights into health status, causes of mortality and causes of MSEs that could not be revealed by any other method.

Stranding patterns

Strandings or beaching of carcasses—dead or alive, single or mass (involving two or more individuals)—can be influenced by many factors, some known and some more speculative. These

include variation in cetacean population density, interannual and seasonal variations in climatic factors like prevailing onshore winds, and coastline length. Drivers of cetacean mortality can also influence stranding patterns, including behaviour and location of commercial fishery operations, high-intensity acoustic activity and disease epizootic events (Murphy 2004, Geraci & Lounsbury 2005, Deaville & Jepson 2011). There are a number of speculative ‘natural’ causes for cetaceans becoming stranded alive, including behavioural tendencies and group cohesion, disease in one or more individuals in a social group leading to some or all of the remainder of the group being stranded alive; unusual environmental conditions, such as electrical storms and other meteorological events; becoming trapped by a receding tide; geomagnetic disturbances and errors in navigation while following geomagnetic contours; confused navigation arising from ‘bathymetric conditions’ (i.e., misleading depth contours); disturbance of echolocation by multiple reflections in bays; pursuing prey too close to shore; and earthquakes (reviewed in Geraci & Lounsbury 2005, Sundarama et al. 2006). However, few MSEs have been forensically investigated. More recently, anthropogenic factors, such as high-intensity acoustic transmission, as used in naval operations, have been increasingly implicated in cetacean MSEs (Jepson et al. 2003, Fernandez et al. 2005, Southall et al. 2006, Weilgart 2007a), including an MSE of short-beaked common dolphins in the UK in 2008 (Jepson et al. in press).

Common dolphins frequently become stranded in the NE Atlantic, especially along the coastlines of Ireland (Berrow & Rogan 1997, Murphy 2004); Britain (Sabin et al. 2002, Jepson 2005, Deaville & Jepson 2011); France (Tregenza & Collet 1998, Van Canneyt et al. 2011); Spain (López et al. 2002); and Portugal (Silva & Sequeira 2003). Recorded strandings of common dolphins have increased since 1990, possibly as a result of increased coastal vigilance, a change in the distribution and abundance of common dolphins, an increase in adverse anthropogenic activities, or a combination of these factors. Annual numbers of strandings have fluctuated in recent years in more northern waters, with similar trends apparent in France and the United Kingdom (see Figure 7; Deaville & Jepson 2011). Strandings have shown a consistent spatial and seasonal pattern, with pronounced winter peaks (ICES WGMME 2005), and for most countries, a high proportion of the common dolphins that became stranded during these winter peaks exhibited external evidence of

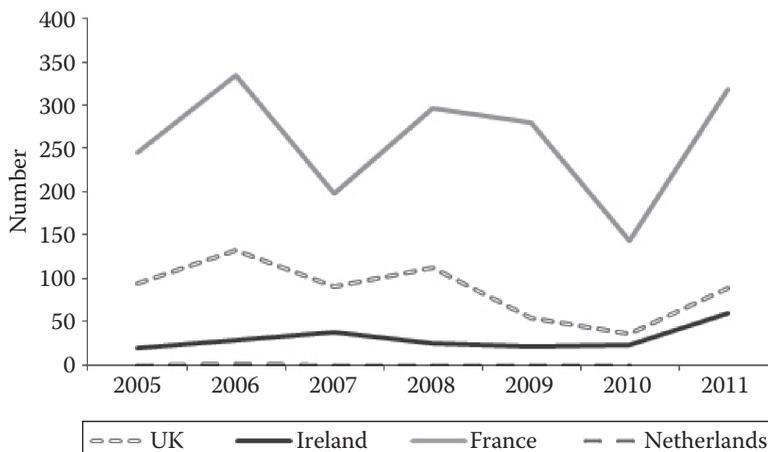


Figure 7 Interannual variation in strandings of short-beaked common dolphins in North-west Europe (2005–2011). One common dolphin became stranded on the Dutch coastline in 2006. (Adapted from Deaville & Jepson 2011. Data provided by the UK Cetacean Strandings Investigation Programme; Irish Whale and Dolphin Group; Centre de Recherche sur les Mammifères Marins, Université de La Rochelle, France; and Naturalis [National Museum of Natural History] in the Netherlands.)

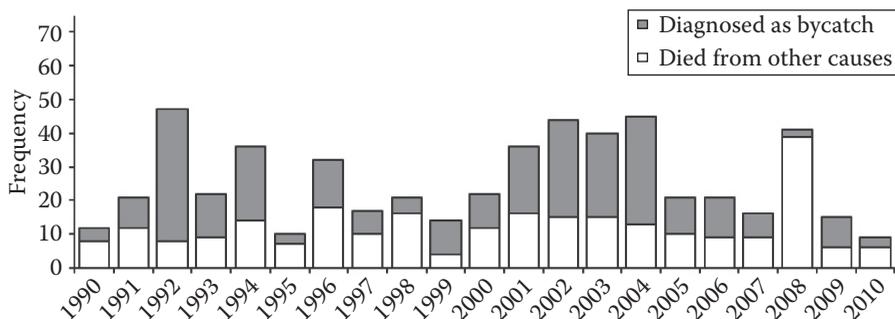


Figure 8 Cause of death for common dolphins necropsied by the UK Cetacean Strandings Investigation Programme ($n = 542$, 1990–2010). Data for 2008 include 26 common dolphins that died during a mass stranding event in Cornwall.

by-catch or pathological evidence of by-catch as the most likely cause of death (Kuiken et al. 1994, Tregenza & Collet 1998, Murphy 2004, Jepson 2005, Deaville & Jepson 2011, Pikesley et al. 2011, Castège et al. 2012, Peltier et al. 2012). Analysis of the age of by-caught dolphins that subsequently became stranded along the Irish, UK, and French coastlines between 1990 and 2006 showed an increased mortality of juveniles, with a peak in mortality of 3-year-olds (Murphy et al. 2007b). It is not established what fisheries or type of fishing nets were involved in these incidental mortalities.

On the Irish coast, the majority of recorded common dolphin strandings have been along the western and southern shores, corresponding with areas of highest sighting rates and direction of prevailing westerly winds (Murphy 2004). Between 1990 and 2003, of all strandings, 49% occurred during the first quarter of the year, and at least 25% of these dolphins were identified as by-catch (Murphy 2004). There was a peak in strandings of 37 common dolphins in 2003 (Brophy et al. 2006); prior to this, the average stranding rate was 11 dolphins per year (Murphy 2004). Peaks in annual numbers stranded (>15 dolphins) resulted from both fisheries interactions and live stranding events (Murphy 2004).

A large number and proportion of common dolphins that became stranded along the UK coastline were diagnosed as by-catch in most years since 1991, mostly between January and April along the SW coast of England (Cornwall and Devon) (Deaville & Jepson 2011). The annual number and by-catch proportion (see Figure 8) of stranded common dolphins in SW England increased in the late 1990s to a peak in 2004 (annual stranding numbers not shown) and then gradually declined thereafter (Jepson 2005, Deaville & Jepson 2011). The reasons for the increase and then reduction in numbers (and proportion) of stranded common dolphins diagnosed as by-catch in SW England around 2004 are not fully understood (Jepson 2005, Deaville & Jepson 2011). In 2011, however, UK stranding numbers were more than twice that of 2010 (see Figure 7).

Since 1989, there were several years with high stranding rates of common dolphin (>250 dolphins per annum) along the French Atlantic coastline, with up to 508 *Delphinus delphis* reported stranded in the year 2000; common dolphins also became stranded at lower frequencies along the French western channel coast (Tregenza & Collet 1998, Van Canneyt et al. 2011). Although there was considerable interannual variability, years of high stranding levels corresponded to events of multiple strandings (ICES WGMME 2005). These events typically occur over periods of 10 to 30 days and involve stranding of predominantly common dolphins, of which about two-thirds show amputation of tail flukes, pectoral flippers or dorsal fin; broken rostrum; or opening of the abdominal cavity—features typical of by-caught animals that have been returned to the sea by fishers (ICES WGMME 2005, Murphy et al. 2007b). The majority of strandings during these peaks are in the southern Bay of Biscay between January and March, and the sex ratio is skewed towards (juvenile) males (ICES WGMME 2005, Murphy et al. 2007b). The fact that juvenile males are more

heavily exposed to adverse interactions with fishing gear may result from differential utilization of space and food (ICES WGMME 2005).

In the southern Bay of Biscay between 1980 and 2002, there was strong seasonality in the at-sea encounter rate of common dolphins (sightings data), strandings, and fishing activity (measured by landings of French fleets), with peaks in all three during the first quarter (Castège et al. 2012). However, the relatively high at-sea encounter rate during summer—a period of fewer fish landings—was not reflected in strandings (Castège et al. 2012). Analysis of drift patterns of cetacean carcasses suggested that as many as 57% and 87% of all stranded common dolphins recorded along the French Atlantic coast originated from the continental shelf of the southern Bay of Biscay within the 100-m and 500-m isobaths, respectively (Peltier et al. 2012). Interestingly, of the tagged dead dolphins released by fisheries in the region, only a very small percentage were actually recovered ashore, and results suggested that approximately 84% of dead cetaceans would sink (Peltier et al. 2012). Peltier et al. (2012) estimated that the numbers of stranded common dolphins reported in winter between 2004 and 2009 (207 individuals on average, of which a minimum of 50% were by-caught) would be about one order of magnitude below the true numbers dying at sea (approximately 2000 individuals per winter, of which at least 1000 would have been by-caught). In recent years, however, as in the UK, the stranding numbers have declined, with 136 common dolphins reported stranded along the Atlantic coast of France (and 8 along the English Channel) in 2010. Of these, at least 60 dolphins showed signs of by-catch in fishing gear (Van Canneyt et al. 2011). Stranding numbers increased again in 2011, though they were still lower than records from the 1990s and early 2000s (Van Canneyt et al. 2012; see Figure 7).

The common dolphin is the most frequently recorded cetacean species in strandings along the Galician coast (López et al. 2002). The annual stranding numbers increased from 1990 to 1999, and as in other areas, there was a peak in strandings during the first quarter (March) (López et al. 2002). This peak coincided broadly with the time of the year when the upwelling index is lowest and winds from the west predominate. Interestingly, a secondary peak was observed in August. Overall, the number of strandings was low in autumn, when easterly winds prevail (López et al. 2002). Between 1990 and 2007, there were 1747 common dolphins stranded along the Galician coastline, of which 606 were 'fresh' enough to assess for evidence of fisheries interactions (Read et al. 2009). There were 146 common dolphins diagnosed as by-catch, with more juveniles than adults recorded and more males than females (López et al. 2002, Read et al. 2009). López et al. (2002) reported significant seasonal differences in average size, with the smallest average body length observed in the first quarter and the largest in the third quarter of the year. The percentage of stranded common dolphins showing signs of fishery interactions has increased over time, from 23% during the 1990s (López et al. 2002) to around 41% in the late 2000s (Read et al. 2009).

Between 1975 and 1998, there were 431 common dolphins stranded along the coast of Portugal (Silva & Sequeira 2003). Interannual variations and pronounced peaks in strandings were reported in some years, and as noted in other studies, annual numbers generally increased during the study period. Highest numbers of strandings occurred from December to April, with a peak in March, primarily along the northern and central coasts. The geographic distribution of strandings was attributed to differences in the distribution or abundance of animals, oceanographic conditions, or bathymetric conditions of the region (Silva & Sequeira 2003). The abundance and spatial distribution of the main prey species (sardines and blue whiting) of the common dolphin off the Portuguese coast corresponded with the distribution of strandings. Significantly more immature individuals were found stranded, and the sex ratio was also biased towards males. Fisheries interactions may have contributed to at least 47% of mortalities in the strandings data (Silva & Sequeira 2003).

Overall, stranding recording programmes in the NE Atlantic show a peak in strandings during the first quarter of the year, stranded individuals are biased towards immature males, and a large proportion of animals show evidence of incidental capture.

Mass mortality events

Mass mortality events have been documented in common dolphins in the NE Atlantic. For example, a common dolphin mass mortality event resulting from by-catch occurred in the United Kingdom in 1992 (Kuiken et al. 1994), though it should be noted that high incidences of by-caught dolphins along the SW coast of England during the winter months are annual events (Jepson 2005, Deaville & Jepson 2011). In 1994, a mass mortality occurred in the Black Sea due to distemper caused by cetacean morbillivirus (CMV) infection (Birkun et al. 1999). This CMV epizootic event in Black Sea common dolphins followed a similar CMV epizootic event of striped dolphins in the Mediterranean Sea between 1990 and 1992 (Kennedy 1998).

Cetacean MSEs are often rather loosely described as two or more individuals of the same species (excluding a cow-calf pair) coming ashore, usually alive, at the same time and place (Geraci & Lounsbury 2005). Common dolphin MSEs have occasionally been recorded in the United Kingdom (Jepson & Deaville 2009), France (Dabin et al. 2008, Viricel et al. 2008) and Ireland (Murphy 2004). A range of causes has been proposed (reviewed in Geraci & Lounsbury 2005), including local topography, presence of shifting sandbanks, exceptionally low tides (Murray & Murphy 2003). More recently, proximity to international naval exercises has been established as the most probable cause of a UK common-dolphin MSE (Jepson & Deaville 2009, Jepson et al. in press).

Infectious diseases

All species coevolve with micro- and macroparasites, and common dolphins are no exception (Gibson et al. 1998). A range of macroparasites was recorded in common dolphins in waters around the United Kingdom (Gibson et al. 1998) and NW Spain (Abollo et al. 1998a), and *Anisakis simplex*-associated gastric ulcers were observed in common dolphins in NW Spain (Abollo et al. 1998b), a condition that has been observed in stranded common dolphins from other countries (e.g., Ireland). Non-specific reactive hepatitis and chronic parasitic cholangitis with lymphoid proliferation have been described in common dolphins stranded in the Canary Islands (Jaber et al. 2003). Pulmonary angiomas was observed in 71% (25/35) of common dolphins stranded in the Canary Islands, which was strongly associated with pulmonary parasitic infestation (Diaz-Delgado et al. 2012).

Microparasites include CMV, which caused the epizootic event in Black Sea common dolphins (Birkun et al. 1999). A novel *Helicobacter* sp. was isolated and characterized from gastric ulcers in both common and Atlantic white-sided dolphins (*Lagenorhynchus acutus*) in the NW Atlantic, suggesting that *Helicobacter* species may play a role in the etiopathogenesis of gastritis and gastric ulcers in cetaceans (Harper et al. 2000). Subsequently, *Helicobacter cetorum* infection has been characterized from gastric ulcers in UK-stranded dolphins, including the common dolphin (Davison et al. unpublished data). Meningoencephalitis associated with *Brucella ceti* infection was also recently reported in a UK-stranded common dolphin (Davison et al. unpublished data), which is consistent with similar lesions more commonly described in striped dolphins in the United Kingdom (Davison et al. 2009). A range of macro- and microparasitic (bacterial and fungal) infections have sometimes been the cause of stranding and death in UK-stranded common dolphins (Jepson 2005, Deaville & Jepson 2011). A study assessing the presence of epidermal virus lesions, morphologically similar to the pox virus, on common dolphins off the Irish coast revealed a low prevalence, with evidence of sexual variation, as twice as many males than females were infected (Murphy 1999).

Non-infectious diseases

A range of traumatic injuries and other causes of death have been diagnosed in common dolphins, including by-catch (Kuiken et al. 1994), boat collision (Deaville & Jepson 2011) and fatal attack

from bottlenose dolphins (interspecies aggression) (Murphy et al. 2005b, Barnett et al. 2009). Tumours are rarely found in common dolphins. The first reported case of a meningioma (tumour) in any cetacean was a microcystic meningioma in a common dolphin stranded on the French Atlantic coast, which was diagnosed by immunohistochemical and ultrastructural analysis (Miclard et al. 2006). A single cavernous hemangioma was found in the lung of one common dolphin in the Canary Islands (Diaz-Delgado et al. 2012). Outside the NE Atlantic, a spontaneous case of renal heterotopia involving the lung was reported in a female adult common dolphin found stranded alive on the northern Adriatic Sea coast of Italy (Di Guardo et al. 2005).

Between 1992 and 2009, a small number of UK-stranded cetaceans, including five common dolphins, were diagnosed with acute and chronic forms of gas embolism (Jepson et al. 2003, Jepson 2005, Deaville & Jepson 2011). The cause of gas embolism is not known but may have a similar mechanism to decompression sickness in humans and experimental animals and be related to excessive supersaturation of tissues with nitrogen on ascent (Jepson et al. 2003, Jepson 2005, Deaville & Jepson 2011). More recently, *in vivo* gas formation has been detected by ultrasound in live-stranded dolphins, including common dolphins stranded in the United States, with off-gassing of supersaturated blood and tissues again considered the most probable origin of the bubbles (Dennison et al. 2011). The close proximity of naval exercises to a mass stranding of beaked whales with acute gas and fat embolic lesions on necropsy indicates a potential role for high-intensity anthropogenic activities (like naval exercises) in the pathogenesis of gas embolism, possibly due to abnormal dive profiles (Jepson et al. 2003, Fernández et al. 2005, Jepson et al. 2005b). Another suspected case of fatal gas embolism was a mature female common dolphin that became stranded on the western coast of Ireland in 2004 (Murphy & Rogan 2004).

Reproductive failure and abnormalities of the reproductive tract

Disturbance at any point of reproduction in marine mammals can lead to failure, evident as abortion, stillbirth, premature birth, or illness or death of the newborn (Geraci & Lounsbury 2009). Reproductive failure can be caused by genetic defects, congenital disorders, nutritional or environmental stress, systemic infection, high levels of certain anthropogenic contaminants/endocrine disruptors and marine biotoxins (Geraci et al. 1999, Reeves et al. 2001, Geraci & Lounsbury 2009). In the NE Atlantic common dolphin population, there has been no reported increase in occurrence of preterm births, stillbirths or early newborn mortality since the year 2000. Within a control group of 'healthy' by-caught common dolphins that subsequently became stranded on the UK coastline, 8.9% of mature females showed evidence of recent miscarriage during their second trimester (Murphy et al. 2012b). There was also an association between high tissue contaminant levels and the incidence of miscarriage (see 'Pollutants' section).

A six-year-old immature female common dolphin found stranded on the SW coast of the United Kingdom was diagnosed as a true hermaphrodite. The individual in question had one ovotestis containing both ovarian follicles and testicular tubular elements and a contralateral ovary (Murphy et al. 2011). This is the first reported case of an ovotestis in a cetacean species, and it is not known if this disorder of genital development is due to abnormalities of genetic or chromosomal origin or inappropriate hormone exposure. A number of other reproductive tract abnormalities have been identified in female *Delphinus delphis* stranded on the UK coastline, which are currently under investigation. These include vaginal calculi and numerous ovarian abnormalities, such as tumours and an ovarian cyst (Murphy et al. 2012b). Other conditions, such as genital warts (Gottschling et al. 2011) and infection of testicular tissue with *Brucella ceti* (Jepson & Deaville 2009), have been reported in males.

The first cases of twins in common dolphins were recorded in the NE Atlantic population. A dead floating female common dolphin, found off the Galician coastline in June 1998, was pregnant with a female fetus measuring 72 cm in body length (with a curved caudal area) located in the left

uterine horn and a slightly deformed male fetus measuring 46 cm in body length located in the uterus (González et al. 1999). A 12-year-old by-caught pregnant female common dolphin, in excellent nutritional condition and health status, died in December 1994 off the UK coast. Both fetuses were male, measuring 27 and 31 cm, and were located in the left uterine horn (Murphy et al. 2012b).

Threats and pressures

Overview of past and present threats

A large number of pressures and threats have the potential to have an impact on common dolphins in the NE Atlantic, the most significant being adverse fisheries interactions. Others include climate change, pollutants, noise pollution and habitat disturbance. Additional possible threats are boat collisions and whale and dolphin watching. Although data are currently being collated on contemporary threats and pressures, there is no information on whether the population has been overexploited in the past or if habitat degradation or loss may have reduced carrying capacity.

Direct fisheries for small cetaceans have operated in the NE Atlantic, though most of the information available on catch rates is only anecdotal. The Portuguese small-cetacean fishery officially operated until 1981, when all cetaceans were given legal protection in Portuguese waters; nonetheless, Silva & Sequeira (2003) noted that common dolphins stranded along the Portuguese coasts until 1998 showed signs of having been hunted. In the 1970s and earlier, bow-riding dolphins were deliberately harpooned for food, and a rough extrapolation of activities of the French fleet fishing in the Celtic and Irish Seas at that time, based on an estimated two dolphins captured per month per vessel, suggests a total of 3000 dolphins (not identified to species) were killed (ICES WGMME 2005). In addition, a retired fisherman from the French tuna lining fleet suggested that possibly 2000 dolphins (not just *Delphinus delphis*) were killed every year, a practice that may also have been undertaken by the Spanish tuna fleet (Antoine 1990, ICES WGMME 2005).

In the 1970s, set nets were not mechanized and were relatively short, and pelagic pair trawling had not yet been developed; thus, it is assumed that incidental capture of dolphins would have been less than following the introduction of new commercial fishing practices (ICES WGMME 2005). However, there may have been indirect effects of fishing activities on the local common dolphin population during the twentieth century through overfishing. Reduced availability of prey caused by overfishing and habitat degradation has been proposed as the main reason for the rapid decline in abundance of common dolphins in the Mediterranean Sea since the 1960s (Bearzi et al. 2003, Cañadas & Hammond 2008). Other putative factors that may also have contributed to the decline include incidental mortality in fishing gear and direct catches, contamination by xenobiotic chemicals resulting in immunosuppression and reproductive impairment, and environmental changes, such as increased water temperatures affecting ecosystem dynamics (Bearzi et al. 2003).

Fisheries interactions

Fisheries affect the NE Atlantic common dolphin population in two different ways: through 'operational effects' and 'biological effects'. Operational effects occur when individuals come into physical contact with fishing gear, which may result in serious injury or death (Northridge 2009). If the marine mammal dies and is subsequently discarded, the process is termed *by-catch* (Read 2008). Operational effects pose a serious threat to many populations of marine mammals due to their slow life histories and limited potential rates of increase (Reilly & Barlow 1986, Read 2008). Fisheries also considerably alter the trophic structure, species assemblages and pathways of energy flow (Pauly et al. 1998, Jackson et al. 2001, Myers & Worm 2003), resulting in ecological changes that may have adverse consequences for cetaceans (Read 2008). These interactions result in biological effects.

Operational effects

Even though fisheries observer programmes offer the potential for spatially explicit, effort-corrected and gear-specific estimates of capture and mortality levels (e.g., Tregenza et al. 1997, Morizur et al. 1999, Lewison et al. 2004, Leeney et al. 2008), there is still a lack of understanding regarding why dolphins are caught in nets. This is hampering scientists' ability to develop effective solutions, and overall, a better understanding of the behavioural interactions of marine mammals with fishing gear is required (ICES Sea Study Group for Bycatch of Protected Species [SGBYC] 2008). Furthermore, the sporadic nature of by-catch makes accurate assessment of its nature and the extent, as well as the development of solutions, problematic (Rihan 2010). Changes in fishing tactics may provide some solutions, such as the use of acoustic deterrent devices (i.e., pingers), enhancing acoustic detectability of nets, or simply avoiding setting nets close to cetaceans or in areas where cetaceans are known to be in high density. Although research has been undertaken on a wide variety of these approaches, to date no definitive mitigation measure has come to the forefront, with each having its own significant concerns. For example, pingers have several issues, including high cost, low resilience and relatively short battery life, potential habituation and displacement effects on marine mammals, and there are still questions about the practicalities of their use in commercial fisheries (Rihan 2010, ICES Working Group on Bycatch of Protected Species [WGBYC] 2012). Nevertheless, in recent years, various pinger types have been found to be effective in mitigating by-catch in different gear types. For example, the acoustic deterrent device DDD02-F has been successfully trialled to reduce common dolphin by-catch in the UK component of the midwater pair trawl fishery for sea bass (*Dicentrarchus labrax*) in the western English Channel, and the effectiveness of a variant in UK gill and tangle net fisheries is being assessed (Northridge & Kingston 2010, Kingston & Northridge 2011).

A large number of fisheries operate in the NE Atlantic, using various different gear types, ranging from towed, static to encircling nets, targeting a wide range of species, including prey species of the common dolphin. Where common dolphins and fisheries target the same fish resource, interactions are inevitable, and these may have negative effects on both the fishery, from an economic point of view, and the common dolphin population, through incidental mortality and resource competition (Brophy et al. 2009). In Australian waters, common dolphins have been observed taking fish directly from the cod end and foraging on discarded fish at the surface (Svane 2005). The latter was also reported by Couperus et al. (1997), who noted that common dolphins were either actively feeding in the vicinity of the pelagic trawl nets targeting horse mackerel (scad) off the SW coast of Ireland or may have been scavenging on discards from this fishery as fresh horse mackerel remains were found in the stomachs of dolphins caught in the trawl. Conversely, common dolphins have been observed, via an underwater video camera system, inside trawls targeting sea bass in the English Channel during winter. It was suggested that individuals may be actively feeding on small, non-target fish inside the net (Northridge et al. 2004). In addition, dolphins left and entered the trawl net at will, as there were sightings and resightings of one or more animals for over an hour (Northridge et al. 2004). Tregenza et al. (1997) suggested that common dolphin may be attracted to gill nets during hauling and shooting, especially when the headline floats strike the steel hoop used to spread the net at the stern of the boat as this produces a loud rhythmical 'tonal clatter'. Although results from a recent UK study further suggested that interactions with gear during shooting or hauling may have more of an effect on by-catch than gear characteristics, by-catch rates of common dolphins in bottom-set nets may also be driven by a temporal and spatial overlap of animals and fishing gear, rather than specific characteristics (e.g., soak time, mesh size) of that gear (Mackay 2011). In contrast, Nielsen et al. (2012) found that harbour porpoises avoided a gill net and appeared to be able to detect it at a distance of more than 80 m. Thus, the incidental capture of porpoises may be due to attention shifts or to auditory masking reducing their ability to detect the gill nets by echolocation (Nielsen et al. 2012).

The EU Council Regulation (EC) No. 812/2004 (European Union 2004) states that independent observations of fishing activities are essential to provide reliable estimates of the incidental catch of cetaceans. It is therefore necessary for

monitoring schemes with independent on-board observers to be set up and for the designation of the fisheries where such monitoring should be given priority. In order to provide representative data on the fisheries concerned, the Member States should design and implement appropriate monitoring programmes for vessels flying their flag engaged in these fisheries. For small-sized fishing vessels less than 15 m overall length, which sometimes are unable to allow an additional person permanently on board as an observer, data on incidental catches of cetaceans should be collected through scientific studies or pilot projects. Common monitoring and reporting tasks also need to be set. (European Union 2004, p. 15)

For vessels less than 15 m, no specified level of precision or coverage or any other guidance on the level of monitoring is given. This has resulted in pilot projects being poorly implemented by some Member States (ICES WGBYC 2011). The regulation also states that the use of acoustic deterrent devices should be required in areas and fisheries with known or foreseeable high levels of by-catch of small cetaceans. However, this only applies to vessels 12 m or over in length.

Currently, monitoring of bottom-set nets in the Celtic shelf and English Channel (ICES subarea VIIId–j) is not required under EU Regulation (EC) 812/2004 as pinger deployment is mandated by the regulation in this area. Therefore, since the introduction of pinger deployment in 2005, limited monitoring has been undertaken by some Member States. This has prevented not only an evaluation of the current rate of by-catch in these fisheries but also an evaluation of the effectiveness of pingers as a mitigation practice. By-catch is not a function of vessel length, and since the majority of European set gill net vessels are less than 15 m in length, controversy has arisen over the implementation of certain requirements for some fishers and not others (ICES WGBYC 2012). However, it should be noted that monitoring of by-catch on vessels under 15 m and measures to mitigate by-catch, if necessary, are mandated by the Habitats Directive.

Table 3 outlines the available incidental capture rates for common dolphins in various NE Atlantic fisheries over a 20-year period (1990 to 2009). Data presented are only for those fisheries where observations were actually recorded, and CVs or percentages of observed fishing effort are not presented. EU Regulation (EC) 812/2004 came into force in 2004, and despite notable improvements in reporting and observer coverage, it is still not fully meeting its objective (ICES WGBYC 2012). For all marine mammals, insufficient sampling in the right fisheries or areas has prevented sound management decisions to be made with respect to cetacean by-catch (ICES WGBYC 2012).

As can be seen in Table 3, however, some fisheries/gear types have higher by-catch rates than others. It should be noted that there are other fisheries for which there are no reliable estimates of by-catch. Even the available estimates should be used with caution as they only provide an incomplete assessment due to low and uneven sampling coverage, with some EU Member States still not fulfilling their monitoring objectives (ICES WGBYC 2011, 2012). Common dolphins are caught incidentally in pelagic trawls, drift nets (surface gill nets), static gear and seine nets, with the highest annual by-catch (2317 dolphins) reported in 2009. Ten years earlier, high rates of by-catch were reported (2101 dolphins in 1999) for the combined Irish, UK and French components of the albacore tuna drift net fishery; this fishery is explored in further detail through the conclusion of this section. Common dolphins have also occurred as by-catch in a number of other fisheries not listed in Table 3, such as Portuguese gill, beach seine and trawl nets (Silva & Sequeira 2003), and Spanish trawls (including ‘very high vertical opening’ bottom pair trawls for hake, *Merluccius merluccius*; ICES WGMME 2005), gill nets, long lines, and seine nets (López et al. 2003). Interviews with fishers from the Galician fleet between 1998 and 1999 suggested an annual by-catch of 200 cetaceans

in inshore waters and around 1500 off shore, with the majority of these animals probably being common dolphins (López et al. 2003).

Monitoring of UK and Irish bottom-set gill net fleets operating in the Celtic Sea targeting hake (pollack, *Pollachius pollachius*, and other gadoids were also caught) between 1992 and 1994 indicated a by-catch rate of 1.4 dolphins per 1000 km of net and a total annual by-catch of 234 (95% CI = 78–702) *Delphinus delphis* (Tregenza et al. 1997). A slightly higher by-catch rate was reported for the UK hake gill net fleet during the period 1999–2000, with most common dolphins caught between October and March (ICES WGMME 2005). In more recent years, high (extrapolated) numbers of common dolphins were caught in hake fishing nets throughout the NE Atlantic. A by-catch estimate of 115 common dolphins per annum was estimated for UK hake set nets in 2008, though higher by-catch estimates were calculated for UK monkfish (230 dolphins) and pollack fisheries (214 dolphins) (Northridge & Kingston 2009, Sea Mammal Research Unit [SMRU] 2009). In the Cornish tangle and gill net fisheries, estimated by-catch rates in 2005–2008 were 1.15 per 100 hauls and 0.36 per 100 hauls, respectively (Northridge & Kingston 2009). The annual by-catch of common dolphins in Irish gill net fisheries for hake and cod (*Gadus morhua*) in the Celtic Sea between 2006 and 2007 was approximately double what it had been in 1992–1994 (Tregenza et al. 1997, Cosgrove & Browne 2007). In addition, all common dolphins recorded in the earlier period were caught in late autumn and winter (Tregenza et al. 1997), a period that was not sampled in the later study, which focused on the maximum effective spacing for acoustic deterrent devices (Cosgrove & Browne 2007). Preliminary data suggest that 773 common dolphins were caught by Spanish hake gill nets in 2009 (ICES WGBYC 2011), and more recently, a by-catch rate of 0.055 common dolphins killed per “fishing trip/haul” was determined for Portuguese polyvalent boats using gill or trammel nets targeting hake and sea bream (ICES WGBYC 2012). By-catch estimates for the whole Portuguese fleet were difficult to ascertain as it is a multigear fishery.

Portuguese purse-seine nets fishing primarily for sardines caught 47 (non-extrapolated number) common dolphins in 2010 (ICES WGBYC 2012), and an incidental capture rate of 0.50 dead common dolphins per haul was determined for the French sardine purse-seine net fishery during the same year (Morizur et al. 2011).

Since 2001, monitoring programmes have primarily focused on pelagic trawl fisheries, through projects such as PETRACET (PElagic TRAWl and CEtaceans) and EU-funded NECESSITY. The PETRACET project monitored annual fishing effort among the main French, Irish, UK, Danish, and Dutch pelagic trawl fisheries in the Celtic Sea and Bay of Biscay between December 2003 and May 2005. Interestingly, no by-catch of common dolphins was observed in mackerel, horse mackerel (scad) or anchovy fisheries (Northridge et al. 2006). The reasons for this are unknown, though these fisheries did differ in temporal and spatial distribution compared to other pelagic trawl fisheries where by-catch was observed (i.e., albacore tuna and sea bass). During the early to mid-1990s, however, common dolphins were reported as by-catch in Dutch horse mackerel pelagic trawl nets fishing off the SW coast of Ireland and French hake pelagic trawl nets in the inner Bay of Biscay (Couperus 1997, Tregenza & Collet 1998, Morizur et al. 1999). Overall, by-catch events observed during the PETRACET project were clumped in both space and time, with 75 dolphins recorded in 13 tows of the French component of the sea bass fishery, of which 8 were recorded in a relatively small area off Brittany (Northridge et al. 2006). Total by-catch for pelagic trawl fisheries (excluding the UK sea bass fishery), where any cetacean mortality was observed, was estimated to be around 620 common dolphins per annum (Northridge et al. 2006). Numbers of common dolphins caught by the European sea bass pair trawl fishery have fluctuated since monitoring began (see Table 3). While the number of by-catch events has declined in the UK fishery, primarily due to effective mitigation and lower fishing effort, by-catch estimates in the French sea bass fishery increased again in 2008 and 2009. Fernández-Contreras et al. (2010) reported common dolphins as by-catch in Spanish pair trawls between March 2001 and December 2003. This fishery was primarily targeting blue whiting (*Micromesistius poutassou*), with mackerel, hake and horse mackerel as secondary target species.

Table 3 Annual estimates of total by-catch of common dolphin *Delphinus delphis* in ICES areas VI, VII and VIII (1990–2009)

Fishery	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009			
Drift nets																							
Irish, UK & French tuna ^a	243	390	608	1347	1580	666	546	947	1706	2101	1589	0	0	0	0	0	0	0	0	0	0	0	
Pelagic trawls																							
French and Irish tuna ^b														133 ¹									
French tuna (ICES VI, VII, VIII) ^{c-e}					95 ²											60	13	120	900				
French sea bass (ICES areas VII and VIII) ^{b-d,f,h}					25 ²									489 ³			290 ⁴	300					
French sea bass (ICES areas VII) ^{e,g}																					40		
French sea bass (ICES areas VIII) ^{e,g}																					300–400		
French pelagic trawl (ICES area VIII) (various species) ^b																					13		
French midwater otter trawl (ICES areas IV, VII, VIII) (bass, horse mackerel, mackerel, herring and sardine) ^f																57							
UK sea bass (ICES IV.1 and ICES IV.2) ^{g,i-l}													190	38	115	439	139	84	50–100 ⁶	7		4 ⁸	
Dutch horse mackerel ^c																							
French hake pelagic trawls ^c																							
Spanish blue whiting ^m																					394	394	

Hauling time, fishing depth (all dolphins were captured during tows made in water shallower than 300 m) and season were identified as the key factors possibly influencing by-catch events.

There were high rates of by-catch in the albacore tuna drift net fishery that operated during the 1990s in the NE Atlantic (Goujon et al. 1993b, Goujon 1996, Harwood et al. 1999). Using landings of albacore tuna as an indicator of effort, a by-catch of 11,723 (CI = 7670–15,776) common dolphins was estimated for the period 1990 to 2000 (Rogan & Mackey 2007). In 1991, the Council of the European Union decided to limit the length of surface gill nets to 2.5 km and in 1997 declared its intention to ban the use of drift nets in the tuna fishery. The ban was implemented in 2002 (European Union 1998, Rogan & Mackey 2007). This resulted in BIM (Bord Iascaigh Mhara, Irish Sea Fisheries Board) and the Irish Marine Institute undertaking tests on experimental trawls to develop alternative fishing tactics. In 1999, over 160 days, 313 hauls were observed. No cetacean by-catch was observed in 90% of hauls, though 125 common dolphins (observed, not an extrapolated estimate) were caught in just four pair trawls (BIM 2000). As noted, this highly clustered pattern of by-catch is not unusual for pelagic trawls. The incidental capture of cetaceans declined between 2002 and 2004 (16 in 2002, 1 in 2003 and 2 in 2004), which may have resulted from the implementation of a number of avoidance techniques by the Irish fleet. These included (1) cessation of fishing when cetaceans were active in the area; (2) extinguishing stern lights while towing at night; and (3) lowering the trawl headline to several metres below the surface. These practices were simple to adopt and did not adversely affect fishing for albacore tuna (BIM 2004). The use of acoustic deterrent devices during 2002 and 2003 may have further reduced cetacean by-catch (BIM 2004). Since 2005, no cetacean by-catch was observed in the Irish albacore tuna pelagic trawl fleet (148 days at sea observed, and 60 of these were by independent observers; R. Cosgrove personal communication, July 2012). Among other things, this was attributed to the use of more powerful sonar, which precluded the need to deploy fishing gear until tuna were reliably detected (ICES WGBYC 2012).

The area of operation of the albacore tuna fishery changed since the introduction of pelagic trawls. The European tuna drift net fishery usually started fishing for tuna in May, north of the Azores, following the migration of juvenile albacore tuna, moving first northwards in June and July and then westwards, to end in September/October along the continental slope off the SW coast of Ireland (Goujon et al. 1993a). In comparison, the pelagic trawl fishery operates in the inner Bay of Biscay along the 1000-m depth contour, up to shallower continental slope waters off the Irish SW coast (BIM 2005). The main reason for the change in fishing location is the larger concentrations of tuna found close to the continental shelf, making it easier to fish with pelagic trawls in this location. Interestingly, the majority of common dolphins incidentally captured by the Irish albacore tuna pelagic trawl fleet between 1998 and 2003 (though primarily in 1998 and 1999) were caught off the SW coast of Ireland (BIM 2005), as had been the case with the Irish drift net fishery (Rogan & Mackey 2007).

Overall, reduction in by-catch in the Irish component of the pelagic trawl fishery was achieved by carefully targeting tuna instead of towing indiscriminately (ICES WGBYC 2012). In contrast, high numbers of common dolphins were caught in French pelagic trawl nets for tuna in 2009 (see Table 3). This was possibly due to difficulties in finding tuna during that year, which may have resulted in some skippers modifying their fishing operations (ICES WGBYC 2012). By-catch again was clustered, with 94% reported in just two trips, involving two pairs of vessels (Morizur et al. 2010). Cetacean by-catch was lower during the following year (Hassani 2012).

Fisheries selectivity of age-sex maturity classes

It is important to identify what age-sex class of individuals is incidentally captured by each fishery in the NE Atlantic. High mortality of mature (especially pregnant) females, calves and individuals approaching maturity will have a more detrimental effect on the common dolphin population than a high mortality rate of mature males. Analysis of by-caught animals in the predominantly winter European sea bass pelagic trawl fishery revealed a predisposition to capturing juvenile and young

adult common dolphins. Of aged common dolphins captured by the French fleet, 85% were less than 11 years of age, and 90% of aged dolphins caught by the UK fleet were less than 13 years, with a reported peak in the age-frequency distribution at 8 and 9 years (Murphy et al. 2007b). These results imply a lack of learned behaviour of juveniles and young mature individuals around nets, whereas mature individuals may have developed suitable behavioural strategies for feeding within trawl nets (Murphy et al. 2007b). Alternatively, some older individuals may not partake in this type of foraging behaviour. A bias towards male common dolphins was observed in nets of Spanish pair trawls targeting blue whiting, mackerel and other species in Galician waters (2001 and 2002), with an average age of 13.4 ± 4.4 (\pm standard deviation, SD) years for male *Delphinus delphis* and 11.5 ± 4.8 years for females (Fernández-Contreras et al. 2010). Two mass capture events comprising only males (7 and 15 dolphins), with an average age of 7.4 ± 3.2 years, were observed in July 2001 (Fernández-Contreras et al. 2010). This further suggests age and sex segregation of the population during summer (Fernández-Contreras et al. 2010).

Low numbers of calves (<1 year old; 3% of the whole aged by-catch sample) and yearlings (6%) were incidentally captured by both the UK and French sea bass pelagic trawl fleets (Murphy et al. 2007b), and no calves were reported in Spanish pair trawls operating off Galicia (Fernández-Contreras et al. 2010). The low by-catch of calves and weaned juveniles may be due to a lack of association of these individuals with trawl nets (i.e., weaned juveniles not actively feeding within the trawl net) or from the cod end. However, the opposite was found for the (summer) Irish albacore tuna drift net fleet, as common dolphins 2 years old or younger or 165 cm long or less comprised 51.2% of the whole by-catch sample obtained between 1996 and 1999, indicating a strong propensity for calves and yearlings to be captured in drift nets (Murphy & Rogan 2006). A large proportion of calves were also reported in the by-catch of French albacore tuna drift nets, which operated in an area extending from 44°N to 51.5°N and from the Bay of Biscay region, 6°W to 21°W (Goujon et al. 1994). It was suggested that a lack of learned behaviour around nets and lower echolocation capabilities in calves were possible causes for their higher capture rate in the tuna drift net fishery compared to other age classes (Murphy & Rogan 2006). In addition, the high mortality rates of calves in drift nets may also have occurred due to a combination of the length of the net (up to 2.5 km), the lack of discriminating behaviour of this gear type (depending on the habitat usage by age-sex maturity groups), and the timing (during the calving season of the common dolphin) and location of the tuna drift net fishery (Murphy et al. 2007b). Sexually mature individuals of both sexes, including pregnant and recently pregnant females, were also incidentally captured by the Irish tuna drift net fleet, with 43% of the 91 aged common dolphins older than 10 years (Murphy & Rogan 2006). Thus, this fishery, which is now banned, was incidentally capturing the most important age-sex maturity groups in the population.

Biological effects

There are two main forms of interaction resulting in biological effects: exploitative competition by fisheries (removing cetaceans' prey) and interference competition, which involves the disruption of cetacean feeding activities as a result of disturbance (Plaganyi & Butterworth 2005). The long-term impacts of fisheries on the NE Atlantic ecosystem are immense, leading to changes in fish communities due to the loss of larger predators and corresponding ecological function (ICES 2008). A number of fish stocks in the NE Atlantic have been overfished, including both pelagic and demersal stocks (Sparholt et al. 2007), and this resulted in a succession of fisheries for species at lower trophic levels, a process known as 'fishing down the food web'. Common dolphins consume an energy-rich diet, and a decline in suitable prey may cause reduced condition and a decline in reproductive output, with extreme cases leading to starvation and death (Certain et al. 2011). Between 1991 and 2010, only 4% (21/537) of common dolphins necropsied by the UK cetacean stranding investigation programme died from starvation (Deville & Jepson 2011).

In the last few decades, there has been awareness that fisheries management should consider the broader impact of fisheries on the ecosystem as a whole and the impact of the ecosystem, and other users of the ecosystem, on fisheries (Food and Agriculture Organization of the United Nations [FAO] 2008). The overall goal is the sustainable use of the whole system, and achieving this goal requires the implementation of an ecosystem approach to fisheries (EAF). This is defined by the FAO (2003) as follows:

An ecosystem approach to fisheries strives to balance diverse societal objectives, by taking account of the knowledge and uncertainties of biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries. (p. 7)

As part of the EAF, ecosystem models representing a wide range of technological and ecological processes affecting species in the ecosystem (including multispecies and whole-ecosystem models) are used to investigate how the system may change under different future scenarios, including different management options (FAO 2008). Lassalle et al. (2012) undertook an ecosystem approach to assess the impact of fisheries on marine top predators in the Bay of Biscay. Although bottlenose dolphins appeared to be sensitive to resource depletion, common dolphins (and harbour porpoises) were most impacted by their incidental capture in fishing gears (Lassalle et al. 2012). However, results further suggested that the Bay of Biscay was not far from overexploitation at the current fishing rate. The Pianka index value for resource overlap with fisheries was high for common dolphins inhabiting neritic waters of the Bay.

Climate change

Common dolphins are wide ranging and have shown a capacity for range expansion. However, the significance of the effects of climate change on the NE Atlantic population is unknown. In general, climate change is regarded as a key threat to all biodiversity and to the structure and function of ecosystems that may already be subject to significant anthropogenic stress (Graham & Harrod 2009). The NE Atlantic has a temperate-to-subarctic climate, and around the United Kingdom and Ireland, common dolphins have been sighted during summer (calving period) in SSTs ranging between 8.1°C and 18.5°C (mean = 14.9°C; SD = 1.6°C, period 1983–1998) (MacLeod et al. 2008). However, this species has been reported from a wide area around the Strait of Gibraltar to Norway (Øien & Hartvedt 2009) and along the mid-Atlantic ridge, with a mean SST of 15.2°C (CI = 12.3–18.1°C; mean depth 3008.8 m, CI = 2145–3872 m; period June 2004; Doksæter et al. 2008). These are considerably warmer temperatures than those reported during the winter in this region. For example, in the western English Channel, where an increased density of individuals has been reported in winter, SST can fall to between 7°C and 10°C (Gislason & Gorsky 2010, Hughes et al. 2011).

The distribution of the common dolphin in the North Sea fluctuated during the twentieth century. Slight distributional shifts into this sea were observed between the 1920s and 1960s and also since the 1990s (see ‘Distribution and abundance’ section). The water temperatures in the North Sea fluctuated during the last century, with a period of low water temperatures between 1950 and 1979 (Lambert et al. 2011). Following this, an abrupt ecosystem shift, or regime shift, occurred in both pelagic and benthic ecosystems of the North Sea (Reid & Edwards 2001, Beaugrand et al. 2008). Increased water temperature was proposed as the primary factor influencing the distribution and increased occurrence of common dolphins off the NW coast of Scotland (period 1992–2003; MacLeod et al. 2005). Seawater temperature in that area has risen 0.2–0.4°C per decade since 1981 (Fisheries Research Services 2003, MacLeod et al. 2005). In addition, the recent summertime incursion of common dolphins into the outer Moray Firth and NE North Sea has been anecdotally attributed to increasing regional sea temperatures (Robinson et al. 2010).

An increase in the observed winter abundance of *Delphinus delphis* in the western English Channel between 1996 and 2006 (MacLeod et al. 2009) coincided with an upturn in reported strandings of the species along the SW coast of the United Kingdom (Jepson 2005, Deaville & Jepson 2011). During this period, there was a 1°C rise in the mean annual SST in the western English Channel (1990–2000), which exceeded any other SST change in the area over the last 100 years (Hawkins et al. 2003). By the 2080s, the temperature of these waters is expected to rise by up to 3°C, which may lead to the loss of some economically important cold-adapted species (Graham & Harrod 2009). In recent years, however, there has been a decline in the numbers of common dolphins becoming stranded along the SW coast of England and the Atlantic coast of France, possibly reflecting other variables, in particular fishing effort (see ‘Stranding patterns’ section).

Although it has been suggested that temperature is a key limiting factor in the northern limit of common dolphins in western European waters, and individuals may shift their distribution to stay within their thermal niche (Lambert et al. 2011), changes in temperature also affect prey species of the common dolphin, influencing physiological and ecological processes in a number of direct, indirect and complex ways (Graham & Harrod 2009). Thus, common dolphins may shift distribution to remain within their ecological niche. The decline in reported strandings off the SW coast of England between the 1930s and 1970s (after an earlier peak in strandings during the 1920s and 30s; see ‘Distribution and abundance’) followed fluctuations in pelagic assemblages (zooplankton and larval fish) in the English Channel during the 1920s and 1930s. Changes were attributed to an increased SST and a reduced Atlantic flow into the Channel—later known as the ‘Russell cycle’—which resulted in a decreased biomass of all higher trophic levels (Southward et al. 2005 and references therein). A decline in the abundance of cold-water fishes in the English Channel was observed at that time (Southward 1963, Evans & Scanlan 1989, Southward et al. 2005), with a northwards shift in their distribution, and it is believed that common dolphins followed (Fraser 1934, Evans & Scanlan 1989, Murphy et al. 2006). In the western English Channel, there has been an alternation in abundance of herring (*Clupea harengus*), a cold-water species, to pilchard (*Sardina pilchardus*), a warm-water species, in response to environmental conditions since at least the fourteenth century. However, the change in fish composition from 1926 to 1936 is seen as a climatically mediated shift exacerbated by intense overfishing, leading to recruitment failure of herring (Southward et al. 2005 and references therein). From the late 1960s onwards, many of the conditions prevailing in the early 1920s returned, along with an increase in common dolphin strandings along the SW coast of England (Evans & Scanlan 1989) and the southern and western coasts of Ireland (Murphy 2004). Since the 1980s, however, conditions in the English Channel have changed again, with warm-water species, such as pilchard, increasing in abundance (Hawkins et al. 2003, Southward et al. 2005 and references therein).

Murphy (2004) linked patterns of common dolphin strandings on the Irish coast (1900–2003) with changing oceanographic conditions due to the North Atlantic Oscillation (NAO). The decline in Irish strandings took place during a negative NAO index phase between the mid-1930s and the mid-1970s (see Figure 9). Following this, there was a sharp reversal to a highly positive NAO index phase (Hurrell et al. 2003), with an associated increase in common dolphin strandings along the southern and western coasts of Ireland (Murphy 2004). Changes in the NAO have had wide-scale effects on the North Atlantic ecosystem, influencing SST and winds—both linked to variation in the production of zooplankton—as well as fluctuations in several important fish stocks across the North Atlantic (Planque & Taylor 1998, O’Brien et al. 2000, Hurrell et al. 2003). Furthermore, the weather conditions associated with the positive phase of the NAO, including an increase in winter storm activity, stronger westerly winds and greater wave heights (Bacon & Carter 1993, Hurrell 1995, Stenseth et al. 2002, Hurrell & Dickson 2004), could also increase the number of strandings of common dolphins directly, not only by driving carcasses ashore but also by contributing to the death of diseased or injured individuals. Recent studies have suggested that variability in the NAO index is due to anthropogenic climate change, and this was an explanation for the intensification (strongly positive) of the NAO up to 1997 (Woollings et al. 2010). This period of intensification was abruptly

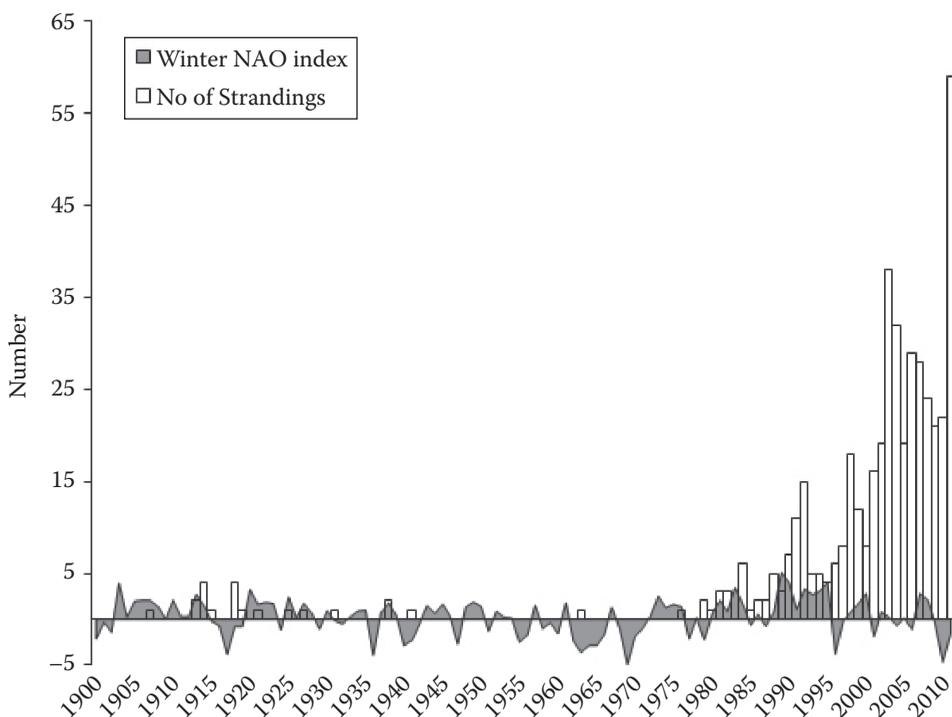


Figure 9 Winter (December–March) North Atlantic Oscillation index and number of common dolphin strandings along the coast of Ireland by year. Winter NAO index based on the difference of normalized sea-level pressure (SLP) between Lisbon, Portugal, and Stykkisholmur/Reykjavik, Iceland, from 1900 to 2010. Data on the winter North Atlantic Oscillation index were obtained from the website of the National Center for Atmospheric Research, USA (Climate and Global Dynamics Division [CGD] 2012). (Adapted from Murphy 2004; updated with strandings data from Berrow et al. 2007, Brophy et al. 2006, O’Connell & Berrow 2012, Philpott et al. 2007 and Philpott & Rogan 2007.)

reversed, and the NAO was weak and variable between 2000 and 2009 due to shifts in atmospheric pressure patterns and then strongly negative in the winters of 2010 and 2011 (Hughes et al. 2012). However, during the 2000s, although the NAO was weak and variable, stranding rates along the Irish coast continued to remain high (>19 per year) (Figure 9). In 2011, strandings of common dolphins were unusually high, with 59 records, the highest in the preceding decade of recorded effort (O’Connell & Berrow 2012). It is not known if this represents a new phenomenon or a temporary peak. Current stranding patterns along the Irish coast may be heavily influenced by anthropogenic activities (see ‘Stranding patterns’ section), which requires further investigation. There may also be other climatic changes in the ecosystem (e.g., Reid & Beaugrand 2012).

Pollutants

Common dolphins are susceptible to the effects of anthropogenic pollutants, such as persistent organic pollutants (POPs), for instance, PCBs, dichlorodiphenylethanes (e.g., dichlorodiphenyltrichloroethane, DDT, a widely used pesticide), hexabromocyclododecane (HBCD), and various heavy metals, such as cadmium and mercury. Pollutants enter the body almost exclusively through the diet, and toxins such as POPs are lipophilic compounds that accumulate in the lipid-rich blubber layers of marine mammals. Apart from some heavy metals, pollutants both biomagnify (higher levels higher up the food chain) and bioaccumulate (increased concentration with age). A large number

of organochlorine compounds (OCs), such as PCBs and DDT, are hormone- or endocrine-disrupting chemicals. Endocrine functions can be altered by these toxins through interference with the synthesis, secretion, transport, binding, action, or elimination of the endogenous natural hormones responsible for homeostasis, reproduction, development, and behaviour (US Environmental Protection Agency [EPA] 1997). As animals can be exposed to a complex mixture of compounds, there may be further significant impacts through additive and synergetic effects.

The production of PCBs and DDT has been limited or completely banned since the 1970s in most developed countries, though DDT is still used in some developing countries for controlling vectors of parasitic diseases (Toft et al. 2004). In addition, OCs, including PCBs, are still being released into the environment through disposal, volatilization of previously released material, and creation of PCBs and dioxins during combustion (Breivik et al. 2002, Katami et al. 2002, Toft et al. 2004).

Persistent organic pollutants

Blubber of common dolphins from the NW coast of Spain, sampled in 1984 and 1996, and of common dolphins entangled in fishing nets in the SW Mediterranean Sea, sampled during 1992–1994, were analysed for organochlorine pollutants (Borrell et al. 2001), which are persistent organic pollutants. Organochlorine levels in both areas were at the mid-to-low end of the range of concentrations detected in common dolphin populations elsewhere in the world and in other delphinids from the same region and were therefore considered unlikely to have played a significant role in the decline of the common dolphin in recent decades in the western Mediterranean Sea.

Retinoids are chemical compounds, related to vitamin A, essential for normal vision, growth, reproduction, immune function, and cellular division and differentiation in mammals. In common dolphins incidentally caught off the NW coast of Spain, age and blubber lipid content were strong determinants of blubber retinoid concentrations in males (Tornero et al. 2006). Retinoid levels were positively correlated with organochlorines in males and negatively in females. As organochlorine pollution levels were only moderate and unlikely to be above proposed threshold levels for mammalian toxicity, the cause-effect relationships between organochlorines and retinoids could not be established (Tornero et al. 2006).

A number of other POPs have been identified in cetaceans, such as butyltins (tributyl tin, TBT; dibutyl tin, DBT; and monobutyl tin, MBT); polyaromatic hydrocarbons (PAHs); and perfluorinated organochemicals. Few studies have measured these in common dolphins in the NE Atlantic region. In the United Kingdom, one of the largest datasets on toxicology in any marine mammal species has been generated on harbour porpoises stranded and incidentally caught between 1990 and 2011 (Jepson 2005, Deaville & Jepson 2011, Law et al. 2012a). The long-term trends in harbour porpoises show stable (and often high) levels of PCBs but declining levels of organochlorine pesticides (such as DDT and dieldrin) (Law et al. 2012a), declining PBDEs (penta-mix brominated diphenyl ether congeners; after an initial increase in the late 1990s) (Law et al. 2010), and only trace levels of butyltins (including TBT) (Law et al. 2012b). Similar trends are likely to be found in common dolphins around UK and northern European waters.

Impacts on reproduction During pregnancy in cetaceans, lipid-soluble contaminants, such as OCs, may be transferred from the mother to the fetus. However, the majority (~80% of OCs) of the pollutant burden accumulated by females (primarily prior to sexual maturity) is believed to be transferred to their firstborn calf during the first 7 weeks of lactation (Cockcroft et al. 1989). In light of this, resting mature females (non-pregnant, non-lactating) with high blubber pollutant burdens and showing signs of recent gravidity may have aborted, or their offspring may have died soon after birth (Murphy et al. 2012a).

An EU-funded Fifth Framework study known as BIOCET investigated the potential impacts of POPs on reproduction in female *Delphinus delphis*. This project pooled samples and data from individuals found stranded in many countries in the NE Atlantic (Ireland, Scotland, France and

Galicia in NW Spain), collected primarily between 2001 and 2003 (Pierce et al. 2008). Factors such as geographic variation in POP burdens in blubber tissue and relationships between POP burdens and age and fatty acid profiles and reproduction were taken into account within the analysis. The most important variable explaining POP profiles in common dolphin blubber was individual feeding history (Pierce et al. 2008). A substantial proportion of individuals in the BIOCET sample had pollutant levels above the threshold of 17 mg kg⁻¹ (PCB lipid weight: mass of PCB per unit mass of lipid; Kannan et al. 2000) reported to have adverse health effects, based on experimental studies of both immunological and reproductive effects in seals, otters, and mink. This threshold was frequently exceeded in common dolphins (40%), especially common dolphins inhabiting waters off the French coast (50%). In addition, pregnant females had lower blubber levels of PCBs and PBDEs than other mature females. This relationship could be interpreted as evidence that high POP concentrations inhibit pregnancy; however, Pierce et al. (2008) pointed out that infertility due to other causes may allow high levels of POPs to bioaccumulate. As data on the health status for all the individuals was unavailable, it is not known if the low estimated pregnancy rate in the BIOCET sample (25%; Learmonth et al. 2004a) was due to disease, loss of nutritional status or high contaminant burdens causing an adverse effect on reproductive output, that is, instigating abortions or infertility.

Subsequent studies by Murphy et al. (2010, 2012a) compared samples and data from the BIOCET project with a control group of 'healthy' common dolphins caught in fishing nets that subsequently became stranded on the SW coast of the United Kingdom between 1992 and 2004. The ovarian corpora (ovulation) scar number significantly increased with PCB burdens in sexually mature *D. delphis* in the BIOCET sample. The majority of individuals with contaminant burdens above the threshold level for adverse health effects were resting mature females (83%) with high numbers of ovarian scars (Murphy et al. 2010). This suggests that (1) due to high contaminant burdens, females may be unable to reproduce and thus continue ovulating; or (2) some females are not reproducing for some other reason, either physical or social, and therefore accumulate higher levels of contaminants (Murphy et al. 2010). Within the BIOCET sample, 92% of mature females with contaminant burdens above the threshold level and corresponding high corpora counts (≥ 15 scars) were obtained from a mass live stranding event in Pleubian, France, in February 2002. As noted, genetic analysis of this group gave no evidence of a matriarchal system, and a lack of genetic relatedness among mature individuals existed (Viricel et al. 2008). Therefore, the existence of non-reproductive females (based on high contaminant loads and high numbers of ovulations) within this social group is noteworthy (Murphy et al. 2010).

The control group of healthy common dolphins also had high PCB burdens, above the threshold level, but these were not inhibiting ovulation, conception or implantation, though the impact on the fetal survival rate required further investigation (Murphy et al. 2010). Studies of the effects of PCBs on reproduction in mink have shown that although ovulation, conception and implantation proceed as normal, fetuses die during gestation or shortly after birth. This results from changes in maternal vasculature in the placenta, leading to degeneration of the trophoblast and fetal vessels and subsequent fetal growth retardation or death (Bäcklin et al. 1997, 1998, Murphy et al. 2012b). A similar situation may be occurring in female common dolphins in the NE Atlantic because within the control group, 8.9% of mature females showed evidence of recent abortion during their second trimester (Murphy et al. 2012b). The association between high contaminant burdens and the incidence of abortion raises serious concerns about the population-level effects of POPs and is currently under further investigation.

Heavy metals

Cetaceans appear to be protected from the effects of many heavy metals due to the presence of metallothioneins as they play a key role in essential metal homeostasis (e.g., Das et al. 2006). Heavy metals accumulate primarily in the liver, kidney and bone. Tissues of fetus-mother pairs of common dolphins stranded along the French coasts (Bay of Biscay and English Channel) were analysed for

their cadmium, copper, mercury, selenium and zinc contents (Lahaye et al. 2007). In the kidneys, fetal cadmium levels were extremely low. Strong relationships between copper and zinc suggested the involvement of metallothioneins since early fetal life. There was only limited maternal transfer of mercury during pregnancy. Hepatic mercury levels in fetuses increased with body length and were also proportionate to maternal hepatic, renal and muscular mercury levels. There was evidence of selenium-based mercury detoxification in both adults and fetuses. Metal levels in muscle, liver, fat tissue and skin were studied in 15 *Delphinus delphis* that became stranded along the Atlantic coastline of Portugal (Carvalho et al. 2002). The concentrations of mercury, tin, chromium, zinc, nickel, cobalt, cadmium, manganese, iron and copper were determined in the liver, kidney and muscle of 24 common dolphins stranded on the Portuguese coast between 1995 and 1998 (Zhou et al. 2001). The concentrations of iron, zinc and mercury were relatively high, particularly in liver, whereas chromium, nickel and cadmium were present at much lower levels or even were undetected. Total mercury concentration increased with body length, and concentrations in the kidney, muscle and particularly liver were higher in females than in males. Total zinc and copper concentrations in muscle decreased with dolphin length (Zhou et al. 2001).

Oil spills

There has been only one detailed study of the effects of oil spills on common dolphins. In December 1999, the tanker ERIKA broke up and sank off the coast of Brittany and continued to leak heavy fuel oil for a few months, impacting the local pelagic and coastal ecosystems. No effects of the ERIKA oil spill were observed on the local common dolphin population (Ridoux et al. 2004). Interestingly, vanadium concentrations in common dolphins were chronically high, both before and after the oil spill (Ridoux et al. 2004).

Noise pollution

Concern regarding the impact of anthropogenically derived sound on marine mammals has been rising in recent decades. The range of sources of anthropogenic noise in the marine environment is wide and varied. Some activities (e.g., shipping and other motorized vessels, use of explosives, drilling, dredging and construction) produce noise indirectly. Other sources, such as active sonars operating at a variety of frequencies, airguns and boomers used in seismic surveys, pingers and acoustic harassment devices, are sources of deliberately introduced sound in the marine environment.

The impact of this noise varies from nil (or attraction, e.g., bow riding) to severe depending on the type, frequency and duration of the noise as well as the relation to the species of concern. Noise can be tolerated, with normal activity patterns maintained and no evidence of an overt response (Würsig & Richardson 2009). Cetaceans may sometimes tolerate noise to remain in a preferred location (e.g., feeding ground) even when the noise is strong enough to cause reactions when the same species is engaged in other activities. In their review of responses of cetaceans to anthropogenic noise, Nowacek et al. (2007) divided responses to noise into three main categories:

- Behavioural responses, such as deviation from normal activity, including changes in swimming speed and breathing/diving activity and avoidance of an area (Richardson et al. 1995). Some effects can be subtle, whilst others are more obvious.
- Acoustic responses: Changes in the type or timing of vocalizations in response to the noise source.
- Physiological responses: Exposure to loud sounds can include temporary or permanent reductions in hearing sensitivity (auditory threshold shifts) (Schlundt et al. 2000).

In addition, physiological effects and symptoms associated with decompression sickness (e.g., embolism and tissue separation), including central nervous system defects (e.g., disorientation,

visual and auditory dysfunction) have been noted (Crum & Mao 1996, Houser et al. 2001). Chronic exposure may also cause stress reactions.

Another issue that has been identified with respect to noise pollution is masking; the noise obscures other sounds of interest to the individual (e.g., feeding clicks). Continuous noise at a similar frequency as the sound of interest is of greatest concern, particularly if the two sounds are received from similar directions (Würsig & Richardson 2009).

Military activity

The impact of military activity and, in particular, use of low- and midfrequency active sonar of high intensity has become a major issue in recent years. A number of MSEs in the last few decades, usually involving beaked whale species, have been temporally and spatially coincident with military activities using such systems (Simmonds & Lopez-Jurado 1991, Todd et al. 1996, Frantzis 1998, Fernández et al. 2005, Rommel et al. 2006, Weilgart 2007a, Mooney et al. 2009, Tyack 2009, Zirbel et al. 2011).

Responses by odontocetes to the use of military sonar include modifications to vocalizations, with the response differing among species. For example, sperm whales (*Physeter macrocephalus*) and beaked whales become silent (Watkins et al. 1993, Cressey 2008), whilst long-finned pilot whales (*Globicephala melas*) increase whistling (Rendell & Gordon 1999), and humpback whales (*Megaptera novaeangliae*) increase the duration of their songs (Miller et al. 2000, Fristrup et al. 2003). Such changes could potentially have an impact on breeding, feeding, and social cohesion, depending on which calls are affected (Weilgart 2007b).

There have been relatively few experimental studies of the effects of military sonars on odontocetes. Mooney et al. (2009) demonstrated both behavioural and physiological effects of midfrequency sonar on bottlenose dolphins. The behavioural reactions were mild, and temporary hearing loss occurred only after prolonged exposures, whereas beaked whales showed a disruption to foraging behaviour (stopped echolocating during deep foraging dives) and avoidance at exposures well below those used by regulators to define disturbance (Tyack et al. 2011).

In 2008, a mass stranding of 26 common dolphins occurred within days of an international military exercise in UK waters. The naval exercise could have caused this large group to come unusually close to the shore, but based on available evidence, the midfrequency sonar in use during the exercise was considered unlikely to have directly triggered the stranding event (Jepson et al. 2009, in press). Subsequent to this mass stranding, the UK Ministry of Defence developed a real-time alert procedure for naval training operations. This enables local information on unusual cetacean sightings, such as the presence of a cetacean group closer to shore than usual, to be incorporated into the training schedule and for operations to be relocated if necessary. This was successfully implemented in April 2009 in relation to the presence of short-beaked common dolphin in the Falmouth Bay area. Over 20 dolphins were seen 15 minutes after Royal Navy sonar trials started. The Royal Navy immediately modified the exercise until the group of dolphins had returned to open water several hours later. Such continual improvement of mitigation strategies by the military themselves is probably the best way to limit future impacts.

Where strandings have been associated with military sonar usage, the major symptoms observed are generally similar to decompression sickness. Exposure to intense, low-frequency sound is thought to stimulate bubble growth within biological tissue, particularly when the tissue is supersaturated with dissolved gas (Crum & Mao 1996). Diving behaviour (depth of dive, depth at which lung collapse occurs and descent/ascent rates) has a significant effect on tissue gas concentration and will therefore influence the susceptibility of a particular species to acoustic exposure (Houser et al. 2001). In addition, the surface interval between multiple dives is related to gas clearance and initial gas tension on subsequent dives (Houser et al. 2001). The species most at risk are those that regularly dive to depths greater than 70 m (the depth at which lung collapse occurs), those with slow descent/ascent rates (slower rates allow greater amount of gas supersaturation) and short surface

intervals (higher initial gas tension on subsequent dives). Bubble growth can cause tissue damage and vascular blockage, which may underlie cetacean stranding events associated with acoustic exposure (Houser et al. 2001) (see ‘Health status and cause of death’ section). Beaked whales in particular are susceptible to such effects (Fernández et al. 2005, Ketten 2005).

Between 1992 and 2004, however, incidences of acute and chronic gas embolic lesions were also identified in five short-beaked common dolphins, four Risso’s dolphins (*Grampus griseus*) and two harbour porpoises in UK waters (Jepson et al. 2005b, Jepson 2005). A decompression-related mechanism involving embolism of intestinal gas or *de novo* gas bubble (emboli) development derived from tissues supersaturated with nitrogen during rapid surfacing was suspected (Jepson et al. 2005b, Jepson 2005). Since these cases, no additional observations of gas emboli were made from animals stranding in UK waters until 2009, when a single case was observed in a Risso’s dolphin (Deaville & Jepson 2011).

Seismic surveys

Oil and gas exploration and production generate a variety of noise, including initial geophysical surveys (using seismic methodologies), rig construction and drilling, and finally structure removal. Of greatest concern is the noise associated with the seismic surveys, which use airguns to generate low-frequency sound. The airguns function by venting high-pressure air into the water column, creating an air-filled cavity that expands and contracts violently, creating sound with each oscillation (Richardson et al. 1995). The sound pulses created depend on the size, number and spacing of the airguns in the array and the air pressure utilized.

Small odontocetes demonstrate the strongest avoidance of seismic survey activity of any cetacean species, with significant increases in fast-swimming activity and declines in sighting rates during periods when airguns are firing (Stone & Tasker 2006, Weir 2008, Gray & Van Waerebeek 2011). Some evidence of temporary threshold shifts has also been noted (Finneran et al. 2002). For common dolphins specifically, avoidance reactions to airgun emissions have been noted in the immediate vicinity, although the species is generally able to tolerate the pulses at 1-km distance from the array (Goold 1996, Goold & Fish 1998).

Operators of seismic activities in many countries are required to work in accordance with specific guidelines. For example, in the United Kingdom, operators are required to follow the Joint Nature Conservation Committee (JNCC) (2010a) guidelines, which include conducting marine mammal observations prior to and during seismic activity and utilizing procedures such as ‘soft start’ (gradually increasing the number of active airguns to allow animals nearby to move away) to reduce and avoid direct harm to animals. Over the years, most recently in 2010, these guidelines have been reviewed and revised in the light of scientific evidence, technical developments and operational understanding.

Aggregate extraction and dredging

The main concern with aggregate extraction is noise generation during survey work. Non-intrusive studies utilize shallow seismic surveys with ‘boomers’, which are considerably quieter than the deep seismic surveys undertaken by the oil and gas industry. Currently, consideration is being given to the possible impact of aggregate extraction works on cetaceans, with a view to guidelines being developed for UK waters. However, by comparison to other anthropogenic sound in the marine environment, aggregate extraction is not considered to be a major threat.

Renewable energy

Marine renewable energy generation is a rapidly evolving industry, with some developments amongst the largest offshore engineering projects ever undertaken. The marine renewables industry encompasses three major sectors: offshore wind, tidal stream and wave energy. The ICES WGMME

assessed the effects of construction and operation of wind farms (2010), tidal devices (2011) and wave energy converters (2012) on marine mammals, work that was synthesized by Murphy et al. (2012b). This section summarizes the main conclusions of the ICES WGMME.

The extraction of energy has many parallels among all three renewable energy sectors, with developments involving placement of substantial structures into the marine environment, requiring large investment and specialized equipment to place and service them. However, there are also fundamental differences when considering the potential interactions with large marine vertebrates, namely, the requirement for submerged moving structures by the ‘wet renewables’ sectors (tidal stream and wave energy).

The majority of offshore wind turbines have monopile foundations, though other foundations, such as tripod, jacket, and gravity, have been used, depending on the seabed type. Monopiles are steel tubes ranging from 2 to 6 m in diameter, driven into the seabed by some thousand strokes of strong hydraulic hammers, produced at a rate of 30–60 pulses per minute (ICES WGMME 2010). Assessment of the effects of the offshore wind industry on marine mammals has focused on near-shore species, such as the harbour porpoise, bottlenose dolphin and seals. To date, pile-driving constitutes the single most important type of impact. Studies have identified a decrease in acoustic activity of harbour porpoises and bottlenose dolphins, up to 20–25 km in one study, following a single pile-driving event (Brandt et al. 2009, Diederichs et al. 2009, Tougaard et al. 2009, Thompson et al. 2010). However, the nature of the behavioural reaction is unknown, as are the consequences of such activities on the long-term survival of individuals (ICES WGMME 2010). During the operational phase, studies have reported either a full recovery of acoustic activity to predisturbance levels (Tougaard et al. 2006) or a significant increase in acoustic activity above baseline levels (Scheidat et al. 2011). In the latter study, increased food availability inside the wind farm (reef effect) or the absence of vessels in an otherwise heavily trafficked part of the North Sea (sheltering effect) were provided as potential reasons for the apparent preference for the wind farm area (Scheidat et al. 2011).

As wet renewable devices are at a relatively early stage of development compared to the offshore wind sector, knowledge of the potential interactions with marine mammals is limited (Murphy et al. 2012a). There are many different concepts (device types) being simultaneously developed within the wet renewables sectors; these devices are extremely diverse in size, shape, method of fixing and many other characteristics (ICES WGMME 2011, 2012). The various designs are at a range of stages of development from conceptual or scale models to a small number of full-scale test rigs deployed at sea (ICES WGMME 2011, 2012). The site requirements for tidal and wave energy extraction are also much more specific than those for offshore wind, as well as being fundamentally different in nature for animals living in these areas compared to other marine areas (Murphy et al. 2012b).

To date, the majority of offshore renewable energy developments within the NE Atlantic have progressed in shallow waters outside the main distributional range of the common dolphin (i.e., Baltic and southern North Seas; Convention for the Protection of the Marine Environment of the North-East Atlantic [OSPAR] Commission 2012), although common dolphins are known occasionally to enter shallow waters. The situation will change in the next few years, however, and the common dolphin population will come into direct contact with the construction and operation of these devices due to the expansion of the industry, both geographically as it moves further off shore and in terms of the number of devices. Many activities of the offshore renewables industry, because of their noise emissions, have been identified as potentially having an effect on marine mammals. These activities may cause hearing damage, disturbance by eliciting behavioural responses, and habitat exclusion (Murphy et al. 2012b). Outlined by Murphy et al. (2012b), these activities include the following:

Site Survey: Noise from seismic surveys, side-scan sonar and survey vessels may cause disturbance or hearing damage.

Construction: Pile-driving may cause hearing damage at close range or disturbance and habitat exclusion. Noise from drilling, dredging, and increased vessel traffic may cause behavioural

change and disturbance. Possible mortality from vessels involved in renewable device installations, especially those using ducted propellers to manoeuvre accurately at small spatial scales. Levels of turbidity (resuspension of sediments) or pollution may increase.

Operation: There are possible collision risks with wet renewable devices. Disturbance and masking of biologically significant signals may result from turbine noise and increased vessel traffic associated with maintenance. Possible disturbance due to habitat alteration, reduced fishing effort (may be positive or negative) and other ecological effects, such as introduction of hard substrate into other environments, may occur.

Decommissioning: Possible disturbance and behavioural change from increased vessel traffic is of concern. Disturbance or hearing damage may result from bottom profiling (seismic surveys and side-scan sonar) or operation of cutting machinery. The use of explosives may cause acute hearing damage or mortality. Levels of pollution may increase.

As for all marine mammals, it is important to assess the cumulative impacts of renewable energy technology on the common dolphin at a population level. This will necessitate a pan-European assessment as common dolphins are highly mobile and are likely to spend only a small proportion of their time within the effective range of a device or even within an array of these devices (Murphy et al. 2012b). It will also become increasingly important to consider the effects of a number of large marine renewable energy sites, with various device types, numbers and configurations of arrays, being constructed relatively close together in space and time. In addition, deployment of marine renewable energy devices is but one of many concurrent activities that might take place within a given marine area; thus, common dolphins in these areas may also be affected by other local anthropogenic activities as well as the large-scale impacts of climate change (Murphy et al. 2012b).

Other impacts

Collisions with vessels and shipping noise

Potentially, all cetaceans can be subject to collisions with vessels, with anecdotal reports occurring for most species (e.g., Kraus 1990, Perry et al. 1999, Knowlton & Kraus 2001, Nowacek et al. 2004). European waters contain some of the busiest shipping routes in the world, such as those in the English Channel and North Sea (Evans et al. 2010). Since the early 1990s, the number of shipping movements, size of vessels and their average speeds have all increased in the region (OSPAR 2010). For European waters, the main areas of collision risk appear to be in parts of the Celtic Sea, Bay of Biscay, and off NW Spain (Evans et al. 2010). Despite this, relatively few cetacean deaths are recorded as a result of ship, small vessel or propeller strikes. Specifically for short-beaked common dolphins, between 2005 and 2010 (inclusive), only 3 of the 129 necropsies undertaken by the UK cetacean strandings investigation programme were diagnosed as ship, small vessel or propeller strike (Deaville & Jepson 2011). An additional four were identified as physical trauma of unknown cause.

Commercial shipping generates intense low-frequency (long-wavelength) noise that can propagate over dozens to hundreds of kilometres (Würsig & Greene 2002). Many factors influence the intensity and frequency of sound produced by vessels, which can lead to different potential effects on cetaceans. Engine type and mounting, exhaust configuration, type of hull construction, power and frequency of sonar units, operation of the vessels (e.g., abrupt changes in speed or gears) and propeller cavitation all affect the noise created. In addition, submarine topography and physical oceanographic factors influence sound propagation and therefore the distance over which the sound can have an impact on cetaceans. The effects of these sounds on cetaceans have not been studied comprehensively, but they may affect communication and other activities associated with sound production and perception (e.g., Castellote et al. 2012). However, small odontocetes are most sensitive to sounds above about 10 kHz. High-frequency hearing is good, with upper limits of sensitive

hearing ranging from about 65 kHz to well above 100 kHz (Richardson et al. 1995), reflecting the use of high-frequency sound pulses for echolocation and moderately high-frequency calls for communication.

The increase over recent years in the recreational use of the sea and activities such as dolphin watching create the potential for an increase in threats through direct physical contact (collisions and propeller damage) and by the sounds introduced into the marine environment.

Whale watching and ecotourism

Few sectors of tourism have experienced the levels of growth in such a short time interval that have been observed in whale- and dolphin-watching operations. Increasingly, concerns have been raised about the impact of such activities on cetaceans (Janik & Thompson 1996, Ananthaswamy 2004, Steckenreuter et al. 2012), but owing to the rapid growth of this sector, the management responses to the potential impacts of cetacean watching have usually been reactive rather than proactive (Lusseau & Higham 2004). Assessing the impacts of tourism on cetaceans is challenging, with few studies attempting to examine more than one aspect of the problem. In addition, few studies have assessed the long-term impact of whale and dolphin watching on cetaceans.

A variety of responses has been observed in marine mammals reacting to tourists, with most studies focusing on changes in behaviour. Common behavioural reactions include schooling animals swimming closer together (e.g., Blane & Jaakson 1994, Nowacek et al. 2001, Steckenreuter et al. 2012); increased swimming speeds (e.g., Blane & Jaakson 1994, Williams et al. 2002a); changes in movement patterns (e.g., Nowacek et al. 2001, Jelinski et al. 2002, Williams et al. 2002b); and changes in resting, feeding, diving and respiratory behaviour (e.g., Baker et al. 1988, Janik & Thompson 1996, Moore & Clarke 2002, Lusseau 2003, Constantine et al. 2004, Dans et al. 2008, Steckenreuter et al. 2012). In addition, changes in habitat use and avoidance of previously preferred areas have been associated with increases in boat traffic (e.g., Glockner-Ferrari & Ferrari 1990, Corkeron 1995, Duffus 1996, Steckenreuter et al. 2012).

A study assessing the impact of tourism on common dolphins in New Zealand revealed that foraging and resting bouts were significantly disrupted (duration and overall time spent in these two states) by boat interactions; the disruptions were to a level that raises concern about the sustainability of this impact (Stockin et al. 2008). Foraging dolphins were more likely to stop foraging and took longer to resume foraging when disturbed by a tour boat compared with periods when a tour boat did not approach.

The presence, density and distance of boats affect cetacean behaviour (Lusseau & Higham 2004). The speed of vessels and their rate of directional change are thought to be critical determinants of the impact of encounters with cetaceans (Nowacek et al. 2001, Williams et al. 2002a,b, Lusseau 2003, Steckenreuter et al. 2012). Unpredictable and erratic vessel movements lead to typical antipredator behaviours in many cetaceans (Lusseau & Higham 2004).

The underwater noise associated with whale-watching operations has been poorly investigated in comparison to noise generated by shipping. Generally, depending on the species concerned, the noise associated with whale-watching vessels is considered likely to mask communication calls and, where large numbers of vessels are continuously present, may cause permanent impairment of hearing (Au & Green 2000, Erbe 2002, Jelinski et al. 2002). There is a growing realization that behavioural changes in marine mammals do occur as a result of engagement with tourist vessels, but the biological significance of such changes has yet to be elucidated (Lusseau & Higham 2004). Behavioural changes may be long term, with impacts on reproductive success and population growth (Janik & Thompson 1996, Notarbartolo-Di-Sciara et al. 2003). If sustainable development of marine ecotourism is to be taken seriously, then rigorous scientific research is required, and the results of such work need to receive action.

With the rapid growth of this industry globally, many nations now have regulations in place that restrict the number of vessels in close proximity to cetaceans and specify minimum approach distances (e.g., Orams 2000, Valentine et al. 2004, Steckenreuter et al. 2012).

Legislation

In the NE Atlantic, common dolphins are covered by a wide variety of legislation, including national, European and international statutes and conventions, all with aims to protect, conserve, manage and study. Although some legislation aims not only to halt deterioration of the ‘status’ of this species but also to achieve a significant and measurable improvement in it, the lack of information on trends in abundance, incidental capture rates, and other parameters prevents a thorough assessment of status for the common dolphin in this region.

International conventions

United Nations Convention on the Law of the Sea

The United Nations Convention on the Law of the Sea (UNCLOS) lays down a comprehensive regime of law and order in the world’s oceans and seas, establishing rules governing all uses of the oceans and their resources (United Nations 2001). It enshrines the notion that all problems of ocean space are closely interrelated and need to be addressed as a whole. The convention governs all aspects of ocean space, including delimitation, environmental regulation, conservation of marine resources, marine scientific research, economic and commercial activities, transfer of technology and the settlement of disputes relating to ocean matters.

Included within the convention (United Nations 2001) are general provisions relating to marine conservation. Specifically, the convention states that contracting parties “shall cooperate with a view to the conservation of marine mammals and in the case of cetaceans shall in particular work through the appropriate international organizations for their conservation, management and study” (p. 48), and that signatories must take measures “necessary to protect and preserve rare or fragile ecosystems as well as the habitat of depleted, threatened or endangered species and other forms of marine life” (p. 101).

Convention on Biological Diversity

The Convention on Biological Diversity (CBD) is one of the three global conventions agreed at the Rio summit in 1992. Its goal is to promote biodiversity, balancing conservation with sustainable use and the sharing of economic benefits that are derived from biodiversity. CBD requires countries to prepare a national biodiversity strategy (or equivalent instrument) and to ensure that this strategy is incorporated into the planning and activities of all those sectors whose activities can have an impact (positive and negative) on biodiversity.

The vision of the CBD Strategic Plan for Biodiversity 2011–2020 is “by 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people” (CBD 2010 Annex paragraph II). As part of these requirements, the European Commission developed and, in 2011, adopted the EU biodiversity strategy (European Commission 2011), a target of which is “to halt the deterioration in the status of all species and habitats covered by EU nature legislation and achieve a significant and measurable improvement in their status so that, by 2020, compared to current assessments, 100% more habitat assessments and 50% more species assessments under the Habitats Directive show an improved conservation status”. As such, it includes the common dolphin.

At the national level, the UK government, for example, launched the UK Biodiversity Action Plan (UK BAP) in 1994. At that time, four plans covering cetaceans were implemented, with a grouped plan for small dolphins, which included the common dolphin. A review of BAP targets was undertaken in 2004, and the Cetacean BAP Steering Group suggested that the United Kingdom should move towards a single Cetacean BAP as many of the targets were generic across all cetacean species, and very few were pertinent to a single species or group of species. This was, however, not implemented, and in 2007 a BAP species and habitat review was undertaken. Under this review, 20 cetacean species were identified, including the common dolphin (JNCC 2010b), for which plans were required. During 2008, priority actions were developed for these species that reflected international obligations. For common dolphins, these are the following:

1. Undertake research on cetaceans using UK waters to identify areas of particular importance for breeding, feeding or migration;
2. Undertake any necessary research and fully implement mitigation measures to reduce by-catch as much as possible;
3. Develop and implement a UK Cetacean Surveillance Strategy;
4. Maintain the UK stranding scheme, which provides an indication of the extent of anthropogenic mortality, and implement appropriate remedial action when necessary; and
5. Undertake research into population structure.

Convention on International Trade in Endangered Species of Wild Fauna and Flora

The Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), also less commonly known as the Washington Convention, aims to regulate international trade in species that are endangered or may become endangered if their exploitation is not controlled (CITES Secretariat 2012). CITES is implemented within Europe through two EC regulations (338/97 and 865/06 as amended). Species covered under CITES are listed in three appendices, with common dolphins listed in Appendix 2. This means that trade in the species is permitted as long as the authorities have ascertained that it will not be detrimental to the survival of the species; that the specimen was not obtained in contravention of the laws of that state for the protection of fauna and flora; and that any living specimen will be so prepared and shipped that it minimizes the risk of injury, damage to health or cruel treatment.

The Bonn Convention and the Agreement on the Conservation of Small Cetaceans of the Baltic, North-East Atlantic, Irish and North Seas

The Convention on Migratory Species (CMS), or Bonn Convention, sets out general provisions for the protection and conservation of certain migratory marine mammals (CMS Secretariat 2012). Common dolphins in the North and Baltic Seas are listed in Appendix II; those in the wider North Atlantic are not listed. Appendix II includes species that have an unfavourable conservation status and that require international agreements for their conservation and management, as well as those that have a conservation status that would significantly benefit from the international cooperation that could be achieved by an international agreement. Parties that are ‘range states’ (countries with waters in the geographical range of the species concerned) of migratory species listed in Appendix II are expected to conclude agreements to benefit the species and should give priority to those species in an unfavourable conservation status. One such agreement is the Agreement on the Conservation of Small Cetaceans in the Baltic, North-East Atlantic, Irish and North Seas (ASCOBANS; ASCOBANS 2012).

ASCOBANS includes a concise Conservation and Management Plan (CMP) that outlines the conservation and management measures to be implemented by signatories. This states that research

“shall be conducted in order to (a) assess the status and seasonal movements of the populations and stocks concerned, (b) locate areas of special importance to their survival, and (c) identify present and potential threats to the different species.” Besides these requirements to monitor abundance and distribution of small cetacean species, the CMP also states that “each party shall endeavour to establish efficient systems for reporting and retrieving bycatches and stranding specimens and to carry out ... full autopsies in order to collect tissues for further studies and reveal possible causes of death and to document food composition”. In addition, the CMP also states that “information shall be provided to the general public in order to ensure support for the aims of the agreement in general and to facilitate the reporting of sightings and strandings in particular; and to fishermen in order to facilitate and promote the reporting of bycatches and the delivery of dead specimens to the extent required for research under the agreement” (p. 8).

Besides the CMP, a number of resolutions have been developed by parties to ASCOBANS, the most relevant of which are the following:

- Resolution 7 of the Fourth Meeting of the Parties in 2003, ‘Cetacean Populations in the ASCOBANS Area’, requires parties to “support further work to elucidate temporal and spatial aspects of distribution of small cetaceans in the ASCOBANS area” (p. 3).
- Resolution 5 of the Fifth Meeting of the Parties in 2006, ‘Incidental Take of Small Cetaceans’, recommends that “total anthropogenic removal is reduced by the Parties to below the threshold of ‘unacceptable interactions’ with the precautionary objective to reduce bycatch to less than 1% of the best available abundance estimate and the general aim to minimise bycatch (i.e., to ultimately reduce to zero)” (p. 1).

To date, ASCOBANS has created and developed conservation plans for harbour porpoises in the Baltic Sea, North Sea, and the Western Baltic, Belt and Kattegat Seas. As yet, no conservation plan has been developed for common dolphins in the NE Atlantic under the auspices of ASCOBANS or any other intergovernmental or non-governmental organization. A conservation plan has been developed for common dolphins in the Mediterranean Sea by ACCOBAMS (the Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic Area).

Convention for the Protection of the Marine Environment of the North-East Atlantic

The Convention for the Protection of the Marine Environment of the North-East Atlantic replaced both the Oslo and Paris Conventions, with the intention of providing a comprehensive and simplified approach to addressing issues associated with maritime pollution; it also provides for conservation and protection of habitats and species. Article 2(1)(a) states the following:

The Contracting Parties shall, in accordance with the provisions of the Convention, take all possible steps to prevent and eliminate pollution and shall take the necessary measures to protect the maritime area against the adverse effects of human activities so as to safeguard human health and to conserve marine ecosystems and, when practicable, restore marine areas which have been adversely affected. (p. 8)

The OSPAR Convention is the mechanism by which 15 governments of the coastal states of NW Europe, together with the European Commission, cooperate to protect the marine environment of the NE Atlantic (OSPAR 2012). Although common dolphins are not listed by OSPAR as a threatened and declining species, the convention clearly states that “definitions of ‘biological diversity’, ‘ecosystem’, and ‘habitat’ are those contained in the Convention of Biological Diversity of 5 June 1992”. Therefore, the OSPAR Convention covers all habitats and species of the NE Atlantic maritime area.

The OSPAR Quality Status Report (QSR) states that to

support an ecosystem approach, OSPAR must extend its focus beyond protecting individual species and habitats or specific sites. Given the array of different actors managing the pressures that impact upon biodiversity and ecosystems, OSPAR should prioritise the development of an effective scheme for monitoring and assessing wider biodiversity status and ecosystem function. This must be linked with the concept of “Good Environmental Status” under the EU Marine Strategy Framework Directive. (OSPAR Commission 2010, p. 141)

Such an approach should benefit common dolphins.

The Bern Convention

The Convention on the Conservation of European Wildlife and Natural Habitats (or the Bern Convention) is a binding international legal instrument in the field of nature conservation that covers most of the natural heritage of the European continent and extends to some states of Africa (European Union 2007). Common dolphins in the North Atlantic are listed in Appendix 2 ‘strictly Protected Fauna Species’, for which the following activities are prohibited:

1. All forms of deliberate capture and keeping and deliberate killing;
2. The deliberate damage to or destruction of breeding or resting sites;
3. The deliberate disturbance of wild fauna, particularly during the period of breeding, rearing and hibernation, insofar as disturbance would be significant in relation to the objectives of this Convention;
4. The deliberate destruction or taking of eggs from the wild or keeping these eggs even if empty;
5. The possession of and internal trade in these animals, alive or dead, including stuffed animals and any readily recognisable part or derivative thereof, where this would contribute to the effectiveness of the provisions of this article.

There is also a requirement for contracting parties to coordinate “efforts for the protection of the migratory species specified in Appendices II and III whose range extends into their territories”.

For Member States of the European Union, the provisions of the Bern Convention are largely taken up in the 1992 Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC), otherwise known as the ‘Habitats Directive’ (see ‘European Legislation’).

International Convention for the Regulation of Whaling

The IWC was set up under the International Convention for the Regulation of Whaling, which was signed in Washington, D.C., in December 1946 (IWC 2012). The purpose of the convention is to “provide for the proper conservation of whale stocks and thus make possible the orderly development of the whaling industry” (p. 1).

The commission has since its inception regulated the catches of the large whale species. Different views, however, are held concerning the legal competence of the IWC to regulate direct and incidental catches of small cetaceans, including the common dolphin. Despite the different views on the question of legal competence, the IWC does recognize the need for further international cooperation to conserve and rebuild depleted stocks of small cetaceans. Each year, the IWC Scientific Committee, through its Sub-Committee on Small Cetaceans, identifies priority species/regions for consideration by a review. Topics considered include distribution, stock structure, abundance, seasonal movements, life history, ecology, and directed and incidental takes. In 2009, the Sub-Committee on Small Cetaceans undertook a worldwide review of the common dolphin (IWC 2009).

European legislation

Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora

The Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC), commonly known as the Habitats Directive, is one of the most important pieces of European legislation aimed at the conservation of wildlife in the European Union. Common dolphins are listed in Annex IV, 'Animal and Plant Species of Community Interest in Need of Strict Protection'.

Article 11 requires Member States to monitor the conservation status of the habitats and species listed in the annexes; Article 17 requires a report on this work to be sent to the European Commission every 6 years. In the directive, conservation status is defined as "the sum of the influences acting on the species that may affect the long-term distribution and abundance of its populations" (Article I(i), p. 5). Conservation status can be considered favourable if

- population dynamics data indicate that the species is maintaining itself on a long-term basis as a viable component of its natural habitats,
- the natural range of the species is neither being reduced nor is likely to be reduced in the foreseeable future, and
- there is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis.

Assessment of favourable conservation status (FCS) therefore requires consideration of range, population, main pressures and threats, habitat and future prospects of the species, including any identifiable trends. These assessments should be undertaken using a standard methodology, designed to facilitate aggregation and comparisons between Member States and biogeographical regions (see the section 'Conservation status' for further information on the FCS assessments of the common dolphin).

Under Article 12:

Member States shall take the requisite measures to establish a system of strict protection for the animal species listed in Annex IV(a) in their natural range, prohibiting: (a) all forms of deliberate capture or killing of specimens of these species in the wild; (b) deliberate disturbance of these species, particularly during the period of breeding, rearing, hibernation and migration; and (d) deterioration or destruction of breeding sites or resting places. (pp. 9–10)

Under Article 12(4) of the Habitats Directive:

Member States should establish a system to monitor the incidental capture and killing of the animal species listed in Annex IV(a), and in the light of the information gathered, Member States shall take further research or conservation measures as required to ensure that incidental capture and killing does not have a significant negative impact on the species concerned. (p. 10)

These apply to common dolphins, with the most significant anthropogenic impact being by-catch.

EC Council Regulation 812/2004 (the 'Fisheries Regulation')

EC Council Regulation 812/2004 (the 'Fisheries Regulation') lays down measures concerning incidental catches of cetaceans in fisheries and amends Regulation (EC) No. 88/98. The measures pertinent to common dolphins in the North Atlantic are the coordinated monitoring of cetacean by-catch through compulsory onboard observers for given fisheries and the mandatory use of acoustic deterrent devices ('pingers') in certain fisheries.

EC Regulation 812/2004 requires that sampling should be such that a by-catch estimate with a CV of less than 0.3 can be achieved. However, for the CV to be determinable, the mean by-catch

rate must be non-zero, so one or more by-catch events must be observed (Northridge & Thomas 2003). In the absence of any observed by-catch, and assuming continued monitoring is needed, the United Kingdom uses the ‘pilot study’ levels of 10% and 5% for the various fishery segments as the most appropriate approach to setting monitoring requirement levels. The United Kingdom is recognized as having one of the best by-catch observer schemes in Europe (European Commission 2009) and secured dispensation from the monitoring requirements in certain fisheries that had been demonstrated to have no by-catch over a 5-year period (e.g., pelagic trawls in the Celtic Sea targeting a variety of species, including mackerel, herring, blue whiting, horse mackerel, sardine, sprat and bass) to focus on those fisheries not covered by the legislation but known or suspected as having high levels of by-catch (Northridge & Kingston 2010) (see ‘Fisheries interactions’ section).

Marine Strategy Framework Directive and Good Environmental Status

The Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC) requires Member States of the European Union to develop marine strategies that apply

an ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services, priority should be given to achieving or maintaining good environmental status in the Community’s marine environment, to continuing its protection and preservation, and to preventing subsequent deterioration. (European Union 2008, p. 20).

To determine good environmental status (GES), 11 qualitative descriptors have been selected, as outlined in Annex I. These cover biological diversity, non-indigenous species, population of commercial fish/shellfish, elements of marine food webs, eutrophication, seafloor integrity, alteration of hydrographical conditions, contaminants, contaminants in fish and seafood for human consumption, marine litter, and introduction of energy, including underwater noise.

In November 2011, OSPAR brought together its contracting parties to discuss proposed indicators and targets of GES for Descriptor 1: Biodiversity. The majority of relevant Member States proposed cetacean indicators and targets associated with abundance and distribution and by-catch. These are currently being further developed through the auspices of ICG-COBAM’s (Intersessional Correspondence Group–Coordination of Biodiversity Assessment and Monitoring) expert group on marine mammals and reptiles with support from the ICES Working Group on Marine Mammal Ecology.

Management of the North-east Atlantic population

Management unit

One common dolphin population exists within the NE Atlantic. As a consequence of a lack of sampling of ‘offshore’ common dolphins for genetic analysis, the actual distributional range of the NE Atlantic population is undetermined. Thus, the ASCOBANS/HELCOM Small Cetacean Population Structure Workshop (Murphy et al. 2009a) and ICES WGMME (2009) recommended that the MU/area for the *Delphinus delphis* population in the NE Atlantic be confined to the continental shelf and slope waters and the oceanic waters of the Bay of Biscay. Taking into account recent abundance estimates, this area was extended to include the surveyed blocks of SCANS-II and CODA (see Figure 4) for the purposes of estimating by-catch limits for this population (Winship et al. 2009). This region also encompasses some of the main locations of commercial fishery operations in the NE Atlantic. As noted in the discussion in the ecological stock section, separate ‘neritic’ and ‘oceanic’ stocks were proposed for the Bay of Biscay (Lahaye et al. 2005). As the sample size of the putative oceanic stock in this study was only 10 individuals, a two-stock approach to management was not proposed by the ASCOBANS/HELCOM workshop (Murphy et al. 2009a) or ICES

WGMME (2009, 2012). A subsequent working paper presented to the IWC Sub-Committee on Small Cetaceans further delineated stock structure within the NE Atlantic using various ecological tracers and proposed that the continental shelf constituted an ecological boundary with some degree of permeability (Caurant et al. 2011). The IWC noted that some links and subunits were questionable due to very small sample sizes, sex bias in sampling, and temporal differences among the samples not being taken into account (IWC 2009). If further analysis supports the designation of oceanic and neritic stocks in the Bay of Biscay, delineation of stock boundaries will prove problematic, primarily as the main concentration of common dolphins in this region during summer (mating period) is along the continental shelf edge and adjacent oceanic waters (SCANS-II 2008, CODA 2009). In addition, the increased abundance of common dolphins on the continental shelf (e.g., western English Channel) during winter suggests onshore movements of individuals (ICES WGMME 2005), thus increasing the available (SCANS-II summertime) abundance estimate for these waters at that time.

Population status

Genetic analysis of stranded and by-caught common dolphins from continental shelf and adjacent oceanic waters of the Bay of Biscay waters revealed a high haplotype diversity of the control region sequences, suggesting a large effective population size within the NE Atlantic (Natoli et al. 2006, Viricel 2006). Combining the SCANS-II abundance estimate for shelf waters (56,221 dolphins, CV = 0.23) and CODA estimate for offshore waters (116,709 dolphins, CV = 0.34) produces an abundance estimate of 172,920, considered to be representative of the abundance of common dolphins for the summer of 2006 (after Winship et al. 2009). Common dolphins are one of the most abundant cetacean species occurring in continental shelf waters of the NE Atlantic, second only to the harbour porpoise with a population abundance of 385,617 (CV = 0.20) individuals in European continental shelf waters (SCANS-II 2008). Some historical abundance estimates exist for common dolphins (see section on population abundance), and 273,159 (CV = 0.26) individuals were estimated for the NASS-west survey block for the year 1995 (Cañadas et al. 2009), a region located outside the MU area.

Worldwide population abundance estimates for the common dolphin vary widely, with 2,963,000 (CV = 0.24) short-beaked common dolphins inhabiting the ETP (encompassing the northern, central, and southern stocks) (Gerrodette & Forcada 2002); 411,211 (CV = 0.21) individuals in waters off California, Oregon and Washington (Carretta et al. 2011); and 120,743 (CV = 0.23) dolphins in the NW Atlantic. The last estimate listed was based on two surveys undertaken in 2004 that had the most complete coverage of the species' habitat (Waring et al. 2011). As no previous abundance estimate exists for the NE Atlantic MU area and the current net productivity rate based on trends in abundance is undetermined, it is not known if the population is currently depleted compared to historical levels. To date, evaluations of the current status of the species have been based on *ad hoc* assessment methods, and as with other cetacean species, in the future it is essential that long-term (decadal) management will encompass, inter alia, appropriate long-term research planning, with the incorporation of a good monitoring programme (Donovan 2005). As part of the improvements in the approach to assessing species status within the NE Atlantic, a collaborative project, the Joint Cetacean Protocol (JCP) has been developed, which should deliver information on the distribution, abundance and population trends of cetacean species. The JCP brings together effort-related cetacean sightings data from a variety of sources, including large-scale international surveys, surveys based on platforms of opportunity, as well as more localized non-governmental data and industry data. These data, collected between 1979 and 2010, represent the largest NE Atlantic cetacean sightings resource ever collated and have been standardized to a common format, checked and corrected (see Paxton & Thomas 2010, Paxton et al. 2011).

Survival can be estimated on the basis of age and reproductive state determined from teeth sections and gonads of stranded and by-caught cetaceans (Murphy et al. 2007a, Mannocci et al. 2012). These parameters can then be used as inputs in demographic models to conduct population projections and risk analyses, as well as potentially providing indicators of population status in their own right (ICES WGMME 2012). However, use of age-at-death distributions for producing life tables, survivorship curves and determining population growth rates has its limitations and biases. It is important when constructing life tables that the sample/population represents a stable age distribution, that is, where the age structure of the population is constant with time, is not growing (or if growing, it is assumed that the rate of growth is constant and known) or subject to any density-dependent processes (Caughly 1966, Barlow & Boveng 1991).

An assessment of available age-at-death data from the NE Atlantic common dolphin population revealed a large sampling bias due to a high immature (and young adult) mortality from anthropogenic activities (Murphy et al. 2007a). Samples obtained directly from fisheries and from stranding projects exhibited this sampling bias—even the sample of stranded individuals for which cause of death was not attributed to incidental capture. However, a time series of such mortality data could provide an indicator of changing population status, with increasing mortality potentially indicating an undesirable trend (ICES WGMME 2012). R_{MAX} is the potential rate of population increase under optimal environmental conditions (i.e., the maximum net productivity rate) and is determined by the intrinsic life-history characteristics of the species (Murphy 2009a). For cetaceans, the default value of maximum net productivity rate, when no specific estimate is available, is 4% per annum. This is based on theoretical modelling showing that cetacean populations may not grow at rates much greater than 4% given the constraints of their reproductive life history (Barlow & Boveng 1991, Waring et al. 2011). Estimates of R_{MAX} for NE Atlantic common dolphins range from 4% to 4.5% per year (Murphy et al. 2007a, Mannocci et al. 2012).

The NE Atlantic common dolphin population exhibits evidence of age-sex segregation (especially outside the mating period), reproductive seasonality, a low pregnancy rate of 26%, extended calving interval of 3–4 years, and a low potential lifetime reproductive output of about 4 calves. Monitoring trends in life-history parameters is an important requirement of both the Habitats Directive and ASCOBANS as this can also be used to assess conservation status (ICES WGMME 2010). Understanding the causes of change is essential for the design and implementation of conservation and management measures, and purely monitoring trends in abundance will not provide this information. With respect to the findings based solely on mortality data, Murphy et al. (2009b) found no evidence of density dependence in reproductive parameters. The low annual pregnancy rate reported throughout the 16-year sampling period may suggest either that the level of anthropogenic mortality did not cause a substantial population decline or that available prey declined at approximately the same rate as the dolphin population. Even if the low pregnancy rate observed in the NE Atlantic is, in fact, close to the natural rate for a common dolphin population in a temperate region, it cannot be ruled out that other factors, such as environmental and anthropogenic activities, including the introduction and release of physical and chemical pollutants, may be contributing to the population's low reproductive output (Murphy 2009b). The impact of pollutants on fetal and newborn survival rates in *Delphinus delphis* is currently being investigated, and preliminary analysis suggests an association between the incidence of abortion and high pollutant burdens (Murphy et al. 2012b).

Summary of main pressures and threats

A number of environmental and anthropogenic factors are considered to affect cetacean population growth rates, including anthropogenic mortality, food availability, disease, pollutants and climate change (Murphy 2009a). The effects of disturbance on cetaceans at the population level are not well understood.

By-catch and entanglement in fishing gear (including discarded gear) are considered main current threats to the NE Atlantic common dolphin population, and even now these effects cannot be quantified due to a lack of data on incidental capture rates in some fisheries and limited sampling in other fisheries. If pollutants have an adverse effect on individual reproductive capabilities, the population would be more vulnerable to other pressures than is normally assumed, especially other anthropogenic activities, such as incidental capture, and would not necessarily recover from these adverse interactions in a predictable way (Murphy 2009b). Within the NE Atlantic, the degree of human disturbance to cetaceans (offshore construction activities, boat traffic), including the level of underwater noise (e.g., seismic surveys, active sonar, piledriving), has been increasing in recent years and will continue to increase due to expansion of the marine renewables industry. Although short, intense noise can cause injury or death to marine organisms, long-term exposure to less-intense sounds can have sublethal effects, including effects related to stress (Tasker et al. 2010). The long-term cumulative effects of increased noise and disturbance on individual cetaceans are currently unknown.

Common dolphins primarily consume energy-rich prey and show both interannual and seasonal dietary variations depending on prey resource availability. The effects of climate change will alter prey species distribution and abundance, which will have a direct effect on the local common dolphin population. In the NE Atlantic, there is evidence of both seasonal movements and long-term distributional patterns in common dolphins, possibly reflecting changes in resource availability. Therefore, it is expected that in the future common dolphins will adapt to effects of climate change; some studies have already shown changes in contemporary distribution and occurrence related to environmental factors (e.g., MacLeod et al. 2005). In the eastern North Pacific, common dolphins have crossed stock boundaries during periods of significant environmental change. A decrease in the abundance of the northern *Delphinus delphis* stock in the ETP and an associated increase off Southern California, starting in the late 1970s, suggest that a large-scale shift in the distribution of common dolphins may have occurred in this region (Anganuzzi & Buckland 1994, Danil & Chivers 2006). There is evidence that *D. delphis* in the southern stock of the ETP moved into the higher-quality habitat of the 'Costa Rica dome' (an area of upwelling, with a shallow thermocline, associated with high biological productivity), which was within the distributional range of the central stock, during the strong El Niño of 1972–1973 (Danil & Chivers 2006). Even if (gradual) changes in environmental conditions and the distribution or density of preferred prey species are not limiting factors, the health status of the NE Atlantic population and ongoing anthropogenic perturbations all affect the population's ability to both adapt to and recover from significant environmental and resource variability (e.g., during periods of abrupt changes in the ecosystem caused by climate, overfishing, or a combination of these) (Beaugrand et al. 2008).

The contemporary NE Atlantic population is exposed to a variety of novel stressors (including chemical, physical and biological). The effect of any given stressor will be conditional on multiple factors, including sex, age and reproductive condition, as well as other stressors currently affecting the individual. The timing of such effects is also important, as the closer an individual is to allostatic overload when subjected to an additional stressor, the more likely it will have an adverse effect (National Research Council [NRC] 2005). Assessing the impacts of these stressors is complex as stressors rarely act independently. Generally, identification of causal relationships between a stressor and a change in survival and reproductive success in individuals has been confounded by the interacting effects of multiple stressors.

Management framework

Human impacts can be classified into two groups that result in either instantaneous or near-instantaneous death (e.g., incidental capture, ship strikes) or those that affect the overall 'fitness' of a population (e.g., pollution, overfishing of prey species and habitat loss) (Donovan 2005). Conservation management is the process of regulating human activities to minimize their negative

impacts on natural environments while allowing sustainable exploitation and development. To maintain species in a favourable conservation status, a comprehensive management framework approach must be developed that clearly outlines management objectives and implements research and monitoring programmes to obtain scientific information necessary to inform management. Two types of management objectives might be set: targets (to achieve certain conditions) and limits (to avoid certain conditions). Objectives can be established both for the state of the population and for impacts on that population (Tasker 2006). The establishment of management objectives and the relative weight given to those objectives (the trade-offs) ultimately require political rather than scientific decisions, though it should be remembered that scientists clearly have an obligation to explain the implications of any decisions by, in part, providing politicians with a range of specific options and their likely consequences (Donovan 2005). Monitoring is a fundamental part of management, and in 2010 the ICES WGMME recommended the adoption of an adaptive monitoring and surveillance approach for cetaceans in the ICES area, under which objectives, monitoring and outcomes are regularly reviewed and updated by a steering group composed of representatives from all relevant stakeholders (ICES WGMME 2010). This requires a coordinated international approach for developing a single assessment for each cetacean species at an appropriate biological scale.

If the European Union is to achieve the goals of its Habitats Directive (i.e., maintaining or restoring FCS for listed species), it will need a comprehensive conservation plan for those individual cetacean species for which conservation issues arise due to anthropogenic causes. Such plans have been produced by ASCOBANS for the harbour porpoise in the North Sea and by ACCOBAMS for common dolphins in the Mediterranean Sea and are based on the available information on status, trends and threats. They identify research needs and set conservation targets to respond to the key threats through threat reduction measures, improved regulations or other mitigation strategies (IWC 2008, Reijnders et al. 2008). If no measure exists for specific threats, it has been recommended that a programme should be established involving stakeholders, with all aspects of mitigation measures considered, including science, practicalities, the legal framework, education and awareness (Reijnders et al. 2008). To date, only measures to mitigate the effects of mortality in fishing gear have been proposed for the NE Atlantic common dolphin population.

Management objectives

Management objectives based solely on detecting trends in abundance are inadequate, since a change in population size does not necessarily signify a change in the optimum sustainable level of that population owing to the fact that the carrying capacity may have changed due to natural causes (Gerrodette & DeMaster 1990). To manage and monitor a population appropriately, both the population condition index and the abundance index need to be assessed to detect demographic changes at an early stage (Gerrodette & DeMaster 1990). The optimum sustainable population (OSP), described by the US Marine Mammal Protection Act, is the number of animals that will result in the maximum productivity of the population or the species, keeping in mind the carrying capacity of the habitat and the health of the ecosystem of which they form a constituent element (Wade 1998). The US National Marine Fisheries Service defined OSP as a population level between carrying capacity K and the population size at maximum net productivity (Gehringer 1976), that is, the maximum net productivity level (MNPL) forms the lower boundary of the OSP range (Gerrodette & DeMaster 1990).

The management goal of the US Marine Mammal Protection Act is to prevent populations from 'depletion' and maintain populations above MNPL, estimated to be between 50% and 85% of carrying capacity (and is more likely to be in the lower portion of that range) (Taylor & DeMaster 1993) or between 50 and 70% of a historic population size thought to represent carrying capacity (Gerrodette & DeMaster 1990, Wade 1998). To estimate the historical population size, at a time prior to the directed (direct fisheries or incidental captures) or indirect (habitat deterioration or

harvest or competition for similar prey) impacts by humans, information on vital rates, numbers of animals killed by humans and a current population abundance estimate are required (Gerrodette & DeMaster 1990).

In European waters, the aim of ASCOBANS is to “restore and/or maintain biological or management stocks of small cetaceans at the level they would reach when there is the lowest possible anthropogenic influence” (ASCOBANS 2000, p. 94), and the contracting parties have agreed that a suitable short-term practical objective is to “restore and/or maintain stocks/populations to 80% or more of carrying capacity” (ASCOBANS 2000, p. 94; 2006). In contrast, the Habitats Directive does not set explicit objectives.

Estimating by-catch limits To enforce management objectives to maintain a species at a certain fraction of carrying capacity and to reduce the negative effects of incidental capture, it is necessary to set by-catch limits for specific species, fisheries and areas. A number of approaches can be used to estimate by-catch limits for the defined MU. Based on the ASCOBANS conservation objective (restoring or maintaining the population at 80% or more of K), a harbour porpoise by-catch limit reference point of 1.7% of population size per year was derived, assuming the maximum annual rate of increase was 4% (IWC 2000). ASCOBANS uses this value for representing ‘unacceptable interactions’, and 1% of the population size is used as an ‘intermediate precautionary objective’ (ASCOBANS 2000, 2006). Although this reference point was originally produced for harbour porpoises, it has subsequently been applied to numerous other cetacean species, including the common dolphin. The limit of 1.7% was derived using a simple deterministic population dynamics model, assuming a single stock with more-or-less independent dynamics. When this is not the case, the limit is liable to be inappropriate (ICES WGMME 2012).

In 2010, the ICES WGMME recommended that we should “move away from implicit and automated conservation targets and towards the explicit definition and justification of target population sizes and management objectives” (p. 146). One aim of the SCANS-II (2008) and CODA (2009) projects was to develop a robust framework using all available information to generate safe by-catch limits for harbour porpoises and common dolphins (Winship et al. 2009). Two candidate management procedures were developed: adaptations of the US potential biological removal (PBR) method, and the IWC’s catch limit algorithm (CLA) (part of the revised management procedure) approach. The PBR procedure takes a single, current estimate of absolute population size as input, while the CLA, a more complicated approach, takes time series of estimates of absolute population size and estimates of absolute by-catch as inputs, and thus should be more conservative. Both procedures, however, explicitly incorporate uncertainty in the estimates of population size (SCANS-II 2008, CODA 2009), unlike the deterministic 1.7% by-catch limit reference point. Three different tunings were developed, ranging from conservative to worst-case scenarios. Before this management procedure can be implemented for a particular species in a particular region, several steps need to be taken. These involve agreement by policy makers on the exact conservation/management objective(s), including what fraction of carrying capacity to maintain or restore populations to, and over what period of time; generation by scientists of by-catch limits for a specified period; and establishment of a feedback mechanism for informing the next phase of implementation of the procedure (SCANS-II 2008).

To use the PBR approach correctly, it has been stipulated that population size estimates should not be older than 8 years (Wade & Angliss 1997) as a population that declines at 10% per year from carrying capacity would be reduced to less than 50% of its original abundance after 8 years (Wade & Angliss 1997). In the NE Atlantic, an abundance estimate for continental shelf waters was determined for July 2005 (SCANS-II) and adjacent offshore waters in July 2007 (CODA). Although large-scale decadal surveys were planned for the NE Atlantic, at present (2013), funding for subsequent repeat large-scale survey(s) of these regions has not been obtained.

Conservation status

To assess conservation status, it is essential to understand population structure, including distribution and abundance, key drivers of population dynamics, key resources and the effects of stressors, especially those caused by anthropogenic interactions.

The International Union for Conservation of Nature (IUCN), founded in 1948, is the world's oldest and largest global environmental organization. IUCN's mission is "to influence, encourage and assist societies throughout the world to conserve the integrity and diversity of nature and to ensure that any use of natural resources is equitable and ecologically sustainable" (<http://www.iucn.org/about/>).

Probably the best known of the IUCN's publications is the Red List of threatened species (IUCN 2012). This has become an increasingly powerful tool in conservation management and decision making worldwide (Rodrigues et al. 2006, Currey et al. 2009, Butchart & Bird 2010). The Red List provides information on population size and trends, geographic range and habitat needs of species. The list also outlines the threats and pressures to which the species is exposed and whether it is considered sacred or whether it is protected by international law.

Short-beaked common dolphins are classified as 'least concern' throughout most of their range "despite ongoing threats to local populations. The species is widespread and very abundant (with a total population in excess of four million), and none of these threats are believed to be resulting in a major global species decline" (Hammond et al. 2008). The common dolphin is classified as 'endangered' in the IUCN Regional Red List for the Mediterranean Sea (Hammond et al. 2008).

Conservation status assessments are also required by Article 17 of the Habitats Directive for European waters (see 'Legislation' section). In 2007, the first FCS reports were submitted by Member States (Figure 10). Status in the combined Marine Atlantic biogeographic region assessment for common dolphins was 'unknown' due to the lack of data on current trends in the population and future prospects. Status in the Mediterranean Sea was considered to be 'unfavourable bad' and in the Macronesia biogeographic region 'unknown (but not favourable)'. The next round of FCS reports is due in 2013.

For the NE Atlantic common dolphin population, historical population size is not known, and due to a lack of data cannot be calculated. Furthermore, a time series of population size estimates for the MU area (see 'Population abundance' section) is not available. There is no information on whether the population has been subject to excessive anthropogenic mortality in the past or if habitat degradation or loss may have reduced carrying capacity. All sightings data used to estimate abundance have been obtained during the summer, whereas pelagic trawl fisheries (and a large number of other fisheries) operate predominantly during the winter. Apart from high rates of incidental captures reported by the tuna drift net fishery during the 1990s, there is a lack of adequate information on by-catch rates in a large number of other fisheries. Even though individual countries are now collecting incidental capture rate data under legislative requirements, there are still large data gaps due to uneven and insufficient sampling of fisheries (see 'Fisheries interactions' section). Consequently, we are unable to estimate reliably an annual population incidental mortality rate. The minimum estimated annual incidental mortality rate for the NE Atlantic MU in 2009 was above the 'intermediate precautionary objective' of ASCOBANS (see Table 3). Large data gaps also exist for other stressors.

Even taking all these unknowns and uncertainties onboard, indicators and targets can be developed to maintain or recover the species to GES in support of the MSFD. These indicators would monitor the impacts of pressures, and ensure that the structure and functions of the ecosystems are safeguarded and are not adversely affected, through coordinated marine monitoring programmes with integrated periodic assessments.

Indicators in support of conservation status assessments

OSPAR's ICG-COBAM proposed a number of potential MSFD indicators for cetaceans, including (1) distribution range and distributional pattern within range, (2) population abundance (detections

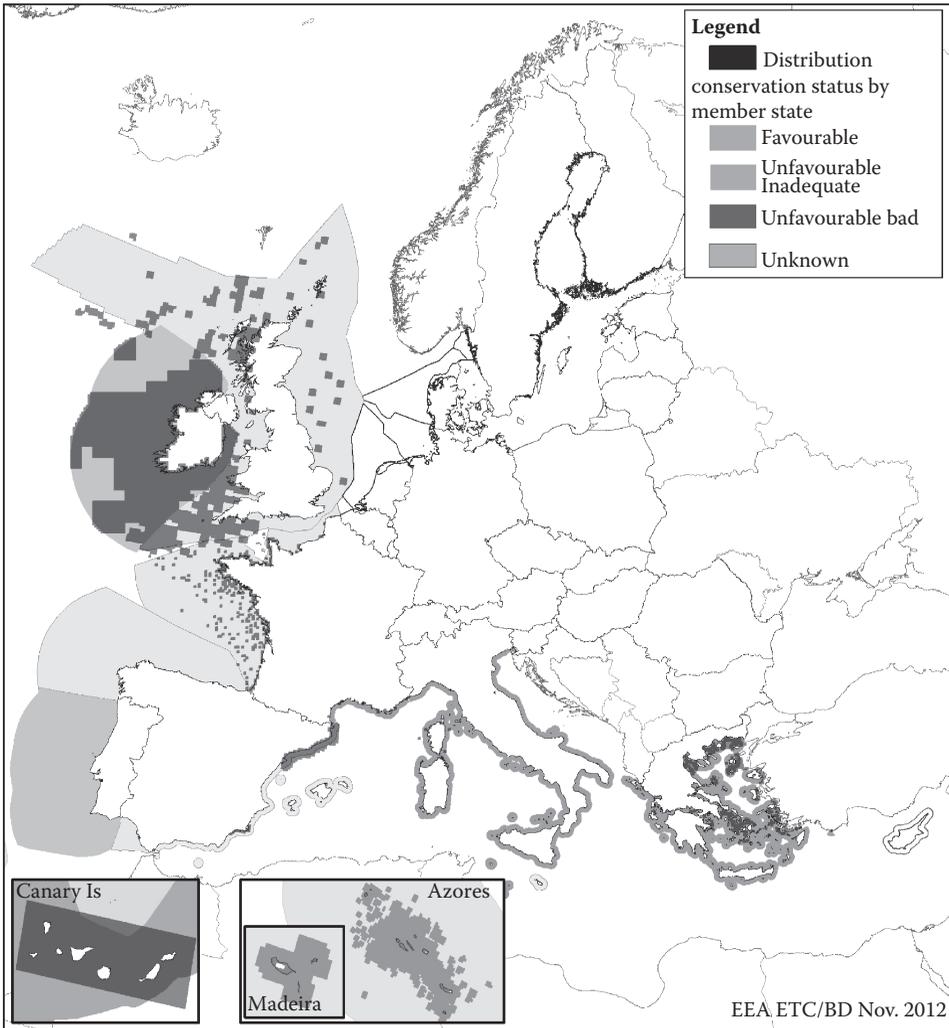


Figure 10 (See also colour figure in the insert) European Union Member States' conservation status assessments for common dolphin in 2007. (Figure produced by Brian Mac Sharry, EEA-European Topic Centre on Biological Diversity [ETC/BD]. With permission.)

of trends in abundance), and (3) mortality rate (anthropogenic mortality in fishing gear) (ICES WGMME 2012). These proposed indicators are largely based on current monitoring requirements for other European legislation. For common dolphins and other cetaceans, unless there is knowledge and continued assessment of population diversity, structure and biological parameters, as well as an understanding of the pressure-state relationships, the usefulness of indicators such as 'population abundance' is questionable, as understanding the root cause of a significant decline in population abundance is necessary for successfully managing that population and achieving GES. Therefore, indicators focusing on pressures and changes in population condition should also be explored. However, it should be noted that data on population abundance are necessary for evaluating other indicators, such as by-catch.

Further development of a by-catch management procedure (based on those developed by SCANS-II and CODA) and creation of a potential MSFD by-catch indicator for common dolphins will be undertaken jointly by the ICES WGMME and WGBYC. For further assessment of

population condition, indicators that use mortality data (e.g., obtained from strandings and fisheries by-catch programmes) can be developed. Two such population condition indicators proposed here are (1) blubber PCB toxicity threshold concentration of 13 mg kg⁻¹ lipid weight (summed ICES-7 chlorinated biphenyl [CB] congener—numbers 28, 52, 101, 118, 138, 153, and 180) and (2) assessing changes in demographic characteristics. Other indicators, not reviewed here, can be obtained from strandings data, such as number of common dolphins whose death was caused by anthropogenic activity.

Blubber PCB toxicity threshold indicator

Detailed research on UK-stranded cetaceans conducted under the UK Cetacean Strandings Investigation Programme has shown strong links between elevated blubber PCB levels and mortality from infectious disease (Jepson & Deaville 1999, Jepson et al. 2005a, Hall et al. 2006) consistent with fatal PCB-induced immunosuppression. In one case-control study of UK-stranded harbour porpoises, the risk of infectious disease mortality increased by 2% for every 1% increase in the summed concentration of 25 CB congeners (Hall et al. 2006). A doubling of risk occurred at approximately 45 mg kg⁻¹ (blubber) lipid. In a second case-control study of UK-stranded harbour porpoises, mean summed 25 CB congeners in the ‘healthy’ control group (death due to physical trauma) was 13.6 mg kg⁻¹, compared with 27.6 mg kg⁻¹ for the animals that died of infectious diseases (Jepson et al. 2005a).

The levels of PCBs in tissues are easily and accurately measured, provided blubber samples from dead stranded animals or biopsies from live animals are available and appropriate sampling and analytical methodologies are in place. Previous studies by Jepson and colleagues (Jepson et al. 2005a, Hall et al. 2006) estimated a threshold of toxicity (including immunosuppression and reproductive impairment) for blubber PCB concentrations in harbour porpoises of 20 mg kg⁻¹ weight (for summed 25 CB congeners). This equates to a blubber PCB toxicity threshold concentration of 13 mg kg⁻¹ (for summed ICES-7 CB congeners) based on standard regressions between summed 25 CBs and summed ICES-7 CBs. This threshold (13 mg kg⁻¹ concentration for summed ICES-7 congeners) could be used for other marine mammal species, including the common dolphin, to assess populations that may show risks of toxic effects at individual and population levels. For target setting, it is recommended that the biological effects from contaminants are kept within safe limits so that there are no significant impacts on, or risks to, marine mammals. The cause-and-effect relationships need to be established and monitored, as well as the impacts of accumulated (independent and interactive) effects. To undertake these tasks, knowledge of information on population growth rates, population structure, life-history parameters and density-dependent changes in these parameters is required. As female cetaceans transfer the majority of their PCB burden to their first calf during pregnancy and lactation (see ‘Ecological stocks’ section), data on age and reproductive status, and whether a female was previously gravid (i.e., pregnant), have to be assessed to provide context to the estimated contaminant burden. For males, age is the most important criterion.

Assessing changes in demographic characteristics indicator

Temporal variations in reproductive parameters can occur due to alterations in the availability of prey resources and population density. Cetacean populations are regulated through density-dependent changes in reproduction and survival, and it has been proposed that food resources are the main causative agent in the expression of density dependence, resulting in an increase in population growth rates (and reproductive output) at low densities (e.g., following large-scale incidental mortality in fishing gear) and a decrease in growth rates (and reproductive output) at high densities (see Murphy et al. 2009b and references therein). However, anthropogenic toxins and disease can alter reproductive rates by decreasing fertility and causing abortions, premature parturition and neonatal mortality (Murphy 2009b).

As part of the various European cetacean stranding programmes, cause of death, health status, nutritional condition and the status of the reproductive tract of individuals are investigated. Teeth, ovaries and testes are collected for subsequent analyses to assess maturity status and age. Strandings following encounters with fishing gear provide the most reliable samples for estimating biological parameters in the NE Atlantic common dolphin population, such as population pregnancy rates, proportion of mature individuals, proportion of females simultaneously pregnant and lactating, average age attained at sexual maturity, nutritional condition, and variations in reproductive parameters with age (see 'Population parameters' section). Dolphins stranded for other reasons, such as old age, disease and parasitic infestation, are likely to be unrepresentative with respect to these parameters (see 'Stranding patterns' section). However, there is evidence that pelagic fishing gear may be somewhat selective with respect to different categories of common dolphin, perhaps due to differences in behaviour, sensory and locomotory capabilities or geographical distribution (see 'Fisheries interactions' section), so the possibility of biases in stranding samples needs to be considered in relation to the parameters of interest when interpreting data from by-catch and fishing-induced strandings.

The conservation target for demographic indicators is no statistically significant deviation from long-term variation. It is important to consider the sample sizes required to detect deviations reliably. For example, pregnancy rate can be readily estimated from stranding samples, but power analysis suggests that changes in the pregnancy rate of the NE Atlantic common dolphin population would need to be extremely large to be detected statistically (Murphy et al. 2009b). With statistical power of 80% and an initial pregnancy rate of 0.26, a sample size of 150 mature females would be required to detect an absolute decrease of 13% or greater (pregnancy rate at 0.13 or below) between two time periods, whereas a sample size of more than 100 mature females would allow detection of a decrease of 15% or greater (Murphy et al. 2009b). With a sample size of only 50 mature females, however, the minimum detectable decrease would be 20%. In contrast, if there was an increase in the pregnancy rate, a sample size of 150 mature females would be needed to detect an increase of 15% or greater with statistical power of 80% (Murphy et al. 2009b). In addition, adequate age and reproductive data from males and females (at least 50 individuals of each sex) are vital for estimating the average age attained at sexual maturity.

Obtaining such large sample sizes of sexually immature and mature individuals is difficult and requires that cetacean stranding and by-catch observer programmes in NW Europe continue to sample all available and suitable carcasses and should be coordinated to standardize procedures. One compromise would be to alter the criterion used for statistically detecting differences. Many managers remain unaware that the standard criterion used for statistical 'significance', the probability of making a 'type I error', $\alpha = 0.05$ (i.e., incorrectly rejecting the null hypothesis in 5% of tests in the long run), is not an objective scientific value but a policy choice based on the most commonly used level of statistical significance (Taylor & Gerrodette 1993).

In addition to these sampling requirements, to interpret reproductive data correctly, population abundance estimates, trends in abundance and data on parameters that affect the dynamics of the population, such as annual mortality rates in fisheries, temporal variations in prey abundance, and levels of anthropogenic toxins, are required.

Recommended research

Numerous different pressures and threats, both anthropogenic and environmental, have affected and will affect common dolphins in the NE Atlantic. Many studies undertaken to date, however, have sampling biases and other limitations, and large uncertainties still exist, primarily due to the difficulties in understanding the true nature and complexity of adaptive responses to stressors, such as noise, on vital functions and rates. In addition, the numerous potential ways in which such multiple and diverse stressors can interact remain poorly understood. Over the next few years, large-scale

expansion of marine infrastructure is foreseen within the NE Atlantic, and when passing through multiple areas with marine infrastructure, dolphins will be exposed to a variety of stressors, varying widely in their nature and impact (ICES WGMME 2010, 2011, 2012). Research is required to support these developments, such that they do not have negative conservation impacts for common dolphins or other cetaceans (as outlined in Murphy et al. 2012b).

Research in areas such as population diversity, structure, abundance and range, seasonal and long-term movements, health status, pollutants, life history, feeding ecology, and mortality rates, including exploring novel causes of death and effective by-catch mitigation practices, should be continued, as should the ongoing evaluation of ecological stocks using tracers that integrate over tens of years (see 'Ecological stocks' section). Novel ways of assessing population range and health status through remote biopsying could be explored (Fossi & Marsili 2011). Finally, with the development of an ecosystem approach to fisheries management, integration of data on common dolphins into ecosystem models not only will allow further elucidation of ecosystem dynamics but also will enable investigation of the effects of climate change, as well as ecosystem and regime shifts, on the local common dolphin population.

Recommended conservation actions

It is crucial that current legal requirements and obligations are fully met and existing management measures implemented and enforced. This could be achieved and aided through the development of an international conservation plan for NE Atlantic common dolphins, which would enable EU Member States to focus on conservation priorities in their waters. Such a plan could be developed through the auspices of ASCOBANS. This plan would continue the identification and evaluation of present and potential threats and reduce potential impacts through the development of threat reduction measures. This conservation plan would accompany other initiatives, such as the ICES surveillance and monitoring framework for marine mammals, and development of MSFD indicators based on population size, distribution, mortality, and population condition.

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