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Abstract The Iberian harbour porpoise (*Phocoena phocoena*) reaches a larger body size than most other harbour porpoise populations and is genetically distinct, albeit closely related to the population in Northwest Africa. Currently comprising an estimated 3000–4000 individuals, genetic evidence and strandings data suggest that the population has declined in recent times, and it is considered to be at risk of extinction. It is distributed all around the Atlantic coast of the Iberian Peninsula, with the highest densities off Galicia in Northwest Spain and Northern and Central Portugal, a highly productive upwelling area characterised by cold-water upwelling. There are occasional reports from the Mediterranean and Macaronesia and some evidence of emigration into the Celtic Sea. It feeds mostly on fish, with pelagic fish being more important than in the diet of porpoises from northern Europe, perhaps due to excursions beyond the narrow continental shelf.

The population faces a number of anthropogenic threats. Historically, porpoises were used for human consumption while current threats include polychlorinated biphenyls (PCBs), with some individuals having concentrations in their blubber above the threshold for impairment of reproduction, and nematode infections, probably also prey depletion, underwater noise and fatal attacks by bottlenose dolphins. The most serious current threat is fishery bycatch mortality. Stranding data suggest that the bycatch mortality increased in the last decade. Although based on information from a small number of documented mortalities (reflecting limited observer coverage especially for small-scale fishing as well as a low number of reported strandings), annual bycatch mortality estimates are in the order of a few hundred animals, which is clearly unsustainable. There is, however, an apparent incompatibility between the high bycatch estimates and the rather similar abundance estimates obtained from large-scale abundance surveys in 2005, 2016 and 2022.

Consistent with population status assessments by Spain and Portugal, OMMEG[1] (Convention for the Protection of the Marine Environment of the North-East Atlantic) concluded that bycatch mortality in Iberian porpoise "is critically exceeding the agreed threshold" of zero. There are several national initiatives in Spain and Portugal including the development of species conservation plans. Continuous reduction of bycatch mortality, preferably until such mortality is eliminated, is a priority to ensure that this population does not disappear in the near future.

Introduction

The Atlantic coast of the Iberian Peninsula (Figure 1) is characterised by a relatively narrow continental shelf, seasonal cold-water coastal upwelling that influences the entire shelf area, equatorward surface circulation in the form of the Canary current, an equatorward Eastern Boundary Current and a poleward flowing slope undercurrent (Fraga 1981, Álvarez-Salgado et al. 2003). Eastern Boundary Currents are shallow, broad equatorward currents, rich in eddies and known for their

[1] OMMEG is the OSPAR Marine Mammal Expert Group, where OSPAR refers to the OSPAR Commission, which administers the Convention for the Protection of the Marine Environment of the North-East Atlantic, better known as the OSPAR Convention (so-called because it originated from a meeting of the Oslo and Paris Commissions), which regulates international cooperation on environmental protection in the North-East Atlantic. It is one of several European Regional Seas Conventions.

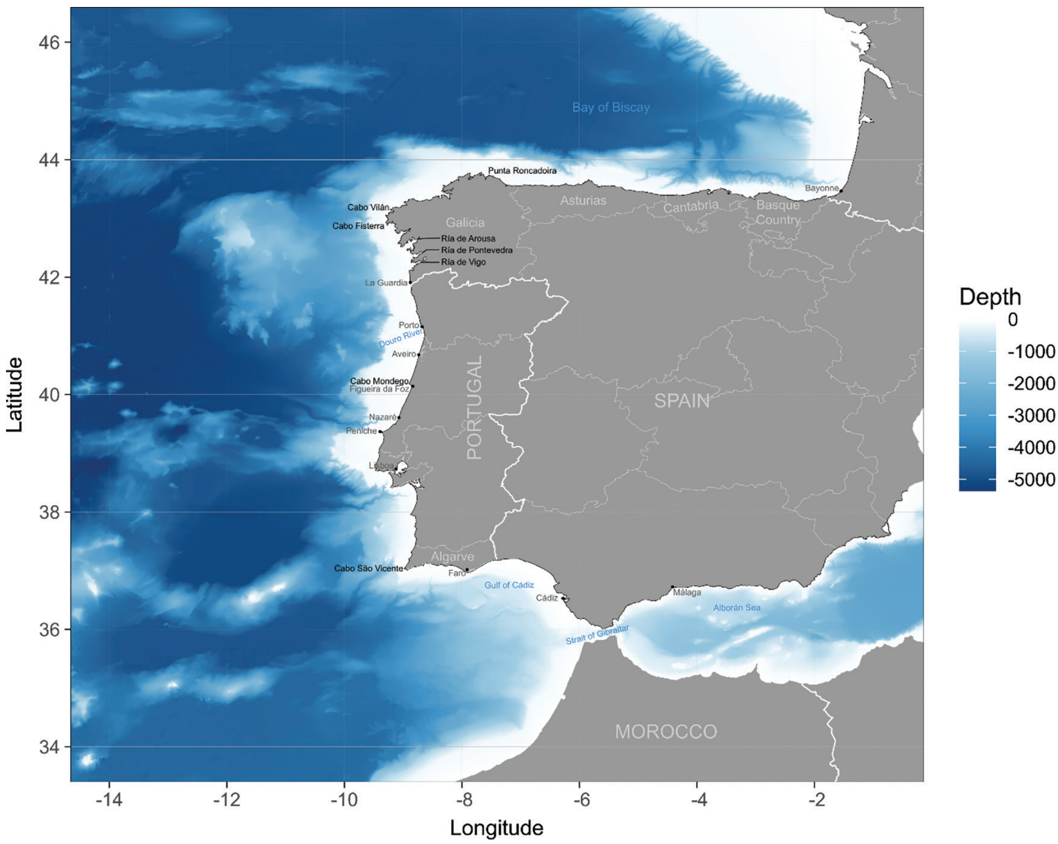

Figure 1 Map of the Iberian Peninsula and adjacent areas showing marine bathymetry data from EMODnet, https://emodnet.ec.europa.eu/en) and the main geographical features mentioned in the text.

upwelling regions along the coast (Moustahfid et al. 2021). The seasonal upwelling supports high productivity, high biodiversity and important fisheries. In Galicia (NW Spain) alone, around 300 species of fish have been recorded (Solórzano et al. 1988), as well as 78 species of cephalopods (Guerra 1992) and at least 22 species of cetaceans and four species of vagrant seals (Fernández de la Cigoña 1990, Penas-Patiño & Piñeiro-Seage 1989, López et al. 2002, López 2017, Covelo & López 2021a). At least two further species are documented from elsewhere on the Spanish mainland Atlantic coast (Fariñas, Petitguyot & Pierce unpublished data). In Portuguese mainland waters, there are at least 150 species of teleost fish (Martins & Carneiro 2018) and 16 species of cephalopods (Jereb et al. 2015) of commercial interest, and 28 species of cetaceans have been reported (Vingada & Eira 2018, Mathias et al. 2023; see also Teixeira 1979, Martin & Walker 1997, Cabral et al. 2006, Wise et al. 2007, Vieira et al. 2009, Brito et al. 2009, Brito & Vieira 2010, Brito & Sousa 2011, Moura et al. 2012, Hammond et al. 2013, Goetz et al. 2015). Mathias et al. (2023) stated that 16 of the 28 cetacean species were commonly found in Portuguese waters while the other 12 were considered as vagrants and that five (vagrant) species of seals are also known from Portugal. Considering records from the extended exclusive economic zone (EEZ) (Correia et al. 2022), the number of cetacean species in Portuguese waters increases to 30.

The harbour porpoise (*Phocoena phocoena*) is one of the most frequently sighted (after the bottlenose dolphin *Tursiops truncatus* and common dolphin *Delphinus delphis*) cetacean species in Atlantic waters of the Iberian Peninsula. In Spanish, the common name for harbour porpoise is *marsopa*; in the Galician language (Galego), it is *toniña* (or *tonina* in López Ferreiro 1895); and in the

Basque Country, it is *moskotxa*. However, in the Canary Islands, *tonina* is a bottlenose dolphin. In Portugal, the harbour porpoise is usually referred to as *boto* (or *bôto*) or *toninha* (or *toninha-comum*). The name *toninha* (sometimes written as *toninha-mansa*) is also used for common dolphin (otherwise *delfim*, *golfinho* or *golfinho-comum*) and (sometimes written as *toninha-brava*) bottlenose dolphin (otherwise *roaz-corvineiro*) (Nobre 1895, Nascimento 1945, Brito et al. 2009, Brito & Vieira 2010, López-Fernández & Martínez-Cedeira 2011, Marçalo et al. 2015, Bencatel et al. 2019). In Porto, fishers used to refer to porpoises as *porcos do rio* (river pigs) (A. Gill, pers. obs.). This list of names and variants may not be exhaustive.

Of the 165 records of *toninhas* in Portugal dating from the thirteenth century onwards compiled by Brito and Vieira (2010), only one can be unambiguously assigned to the harbour porpoise – an animal bycaught and photographed in 1977. It should be noted that the English word 'porpoise' has been used somewhat indiscriminately in the recent past to indicate any small cetacean. For example, Coe et al. (1984) used the word "porpoise" when describing techniques for releasing spotted, spinner and common dolphins (*Stenella attenuata*, *Stenella longirostris*, and *D. delphis*) from purse seine nets in the Eastern Tropical Pacific tuna fishery.

In the mid- to late nineteenth century, several authors referred to the presence of harbour porpoise in Northwest Spain and in Portugal. López Seoane (1861) commented that it could be seen all year round in Galicia, appearing in the ports in summer. Graells (1870) referred to its presence along the whole coast of Galicia and Cantabria, noting that it sometimes entered into the ports and stating that he was unable to obtain a specimen as the fishermen did not target them, only occasionally catching them accidentally in their nets (and resulting in damage to the nets and escape of the catches). Du Bocage (1863) referred to both porpoises ("le Marsouin (*Phocœna communis*)") and common dolphins ("le Dauphin (*Delphinus delphis*, L.)"), especially the former, as usually being seen associated with rivers in Portugal (sometimes at a considerable distance from the river mouth): "se montrent habituellement dans nos principaux fleuves jusqu'à une distance plus ou moins grande de leur embouchure". Nobre (1895) referred to *Phocaena communis* in the Algarve region, southern Portugal, noting that the common name was toninha whereas *Delphinus delphis* was usually referred to as golfinho, and said he saw porpoises frequently while travelling between Lagos and Sines: "je l'ai vu en abondance au nord du cap S. Vincent, pendant la traversée de Lagos à Sines". Both these records clearly distinguished between porpoise and common dolphin. Despite the existence of such records, much of the available information on porpoise occurrence in the Iberian Peninsula derives from the last 50 years, and evidence of significant decline was already available in the 1980s.

Sequeira and Ferreira (1994) noted that while the early naturalists (in the late nineteenth and early twentieth centuries, including the above-mentioned du Bocage and Nobre) considered porpoises to be very common along the Portuguese coast, the situation had "changed drastically" since then, with a reduction in sightings of larger groups and a decline in strandings between 1977 and the late 1980s (see also Sequeira & Teixeira 1988). These authors also noted that many of the stranded animals during this period had net marks around the head and flippers. Perez and Nores (1988) reported that the harbour porpoise declined in Asturias (northern Spain) between 1977–1983 and 1984–1987, based on data from strandings, sightings and catches. They also noted that 7% of 39 bycaught cetaceans obtained during 1977–1987 were porpoises, caught mainly in fixed nets and purse seines.

Lens (1997) summarised almost 200 records from strandings, bycatch and sightings of harbour porpoise, around 100 of which referred to strandings. He observed that porpoise was most abundant along the west coast of Spain, noting that it was absent from the western Mediterranean, and may have declined in the Bay of Biscay and disappeared from the Canary and Azores islands. Over the two and a half decades since that review, new information on Iberian porpoises has emerged from ongoing monitoring of cetacean strandings and fishery bycatch mortality, regional sightings surveys, and various national and international projects. Monitoring of strandings and research on stranded animals have been a major source of information, especially since the establishment of stranding networks in Galicia (run by Coordinadora para o Estudo dos Mamiferos Mariños (CEMMA)) in 1992 and along the central and northern Portuguese coast (run by Sociedade Portuguesa de Vida Selvagem (SPVS) in 2000. Several other networks now cover much of the remaining Atlantic mainland coast and a large part of the coastline of Macaronesian islands (i.e. the Azores, Canaries and Madeira archipelagos). Several large-scale cetacean sightings surveys over the last two decades included Atlantic waters of the Iberian Peninsula, namely, the Small Cetaceans in European Atlantic waters and the North Sea (SCANS II, III and IV) surveys, in 2005, 2016 and 2022, and the Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA) survey in 2007 (Hammond et al. 2009, 2013, 2017, 2021, Gilles et al. 2023). Additional information on distribution and abundance derives from numerous regional and local surveys, conducted from boats, aircraft and land-based observation points (e.g. Pierce et al. 2010, Llavona Vallina 2018, Torres-Pereira et al. 2022).

The biology and ecology of the harbour porpoise are generally well documented on both sides of the North Atlantic (e.g. Sørensen & Kinze 1994, Lockyer 1995a, Read & Hohn 1995, Haug et al. 2003 (and chapters therein), Santos & Pierce 2003, Learmonth et al. 2014; see also numerous reports[2] by the US National Oceanic and Atmospheric Administration). However, the species is less well studied in the Iberian Peninsula, in part because more frequently sighted and stranded species such as common dolphin and bottlenose dolphin (e.g. López et al. 2002, Pierce et al. 2010, Saavedra et al. 2018) have received more attention. Factoring in the small population size, much of the recent evidence available about the biology, ecology and status of Iberian porpoises derives from observations of/on small numbers of animals.

Research on population genetics during the last two decades has revealed that the Iberian porpoise constitutes a genetically distinct population, most closely related to the porpoises of West Africa, and discussions about raising it to subspecies status are ongoing (Fontaine et al. 2007, 2010, 2014, Fontaine 2016, Chehida et al. 2021; see Taxonomy, genetic status, and phylogeography of the Iberian population). The precise limits of its distribution are unknown, and for this reason, within this review we refer to (the small number of) records from the western Mediterranean and Macaronesian islands. The population is small, probably comprising no more than 3000 – 4000 animals and possibly considerably fewer (see Distribution and abundance). Mortality caused by fishery bycatch is of particular concern (Read et al. 2020, Carlén et al. 2021, Pierce et al. 2022, Celemín et al. 2023, Torres-Pereira et al. 2023a). The International Council for the Exploration of the Sea (ICES) Ecosystem Overview for the Bay of Biscay and Iberian Coast states that harbour porpoises "are being caught as bycatch off Iberia, mainly in set nets and beach seines, to the extent that may affect sustainability of the local population" (ICES 2021b).

Despite theoretically strict legal protection (see section 'Conservation of porpoises in Europe: Legal Protection and its implementation'), the conservation of harbour porpoises in Europe is arguably failing due to a combination of "public disinterest, lack of political will to implement conservation measures, and complicated fishing-related issues" (Carlén et al. 2021). A workshop organised by the North Atlantic Marine Mammal Commission (NAMMCO) in 2018 reviewed the status of European porpoise populations, assessed the safe limits for fishery bycatch mortality and compared these with estimated bycatch mortality. The approach taken was based on Potential Biological Removal (PBR, Wade 1998), which is used routinely in the USA for the implementation of the Marine Mammal Protection Act, and fitting logistic population growth models on abundance data within a Bayesian framework (Zerbini et al. 2011). The results highlighted the vulnerable status of the Baltic Proper subpopulation and the Iberian population. In the latter case, based on the best estimates of population size and applying a recovery factor of 0.5, the annual PBR was estimated to be 25 animals (North Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research 2019). Although a subsequent workshop (NAMMCO 2019) concluded that, due to data limitations, not all assessments were reliable for generating management advice, the small size of the Iberian population and the Baltic Proper subpopulation means that a more optimistic view of their status is unlikely to emerge from future calculations.

[2] See https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-species-stock#cetaceans---porpoises

Concerns about cetacean bycatch in Europe have received greater attention since 2019, when 26 European non-governmental organisations (NGOs) called for the implementation of Fishery Emergency Measures (under the Fisheries Common Policy) to reduce bycatch mortality of common dolphin in the Bay of Biscay and harbour porpoise in the Baltic Sea (Butler-Stroud & Rouley 2019). The European Commission requested advice from ICES concerning the amount of bycatch mortality and appropriate measures to reduce it (see ICES 2020a,b). ICES (2021a) undertook a further assessment of bycatch mortality in response to a request from the OSPAR Commission. The NGOs had considered submitting an Emergency Measures document for the Iberian porpoise but concluded there was insufficient information available (S.J. Dolman, Pers. Comm.).

The Convention on Migratory Species adopted a Concerted Action for the harbour porpoise in the Baltic Sea and the Iberian Peninsula, as proposed by Coalition Clean Baltic, Whale and Dolphin Conservation, Humane Society International and ORCA, referring to the Iberian porpoise as a "critically isolated population", in 2020 (CMS 2020). The Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS), through its Secretariat, and the International Whaling Commission's Scientific Committee (IWC SC) (IWC Scientific Committee 2021, 2022, ASCOBANS 2021) have both recently expressed concerns about the status of the Iberian harbour porpoise population. The 26th Meeting of the Advisory Committee of ASCOBANS in 2021 discussed a draft proposal to include the Iberian harbour porpoise population in the Appendices of the UN Convention on the Conservation of Migratory Species of Wild Animals (CMS or the Bonn Convention), noting that this population "has been identified as a high priority for conservation due to its low abundance, genetic distinctiveness, low genetic diversity, and evidence of unsustainably high levels of bycatch mortality in fishing gear" (ASCOBANS 2021). In its 2022 report, the IWC SC recommended "immediate actions to effectively reduce, and where possible eliminate, bycatch of harbour porpoise throughout Iberian Peninsula waters", noting that "measures are urgently needed for fisheries using gillnets and trammel nets but also for beach seines along the Portuguese coast which are used in some areas of high porpoise density" (IWC Scientific Committee 2022). The OSPAR Marine Mammal Expert Group (OMMEG) suggested that a bycatch limit of zero should be applied to the Iberian porpoise population (see ICES 2021a). This threshold was agreed by contracting parties at the Biodiversity Committee of OSPAR in autumn 2021.

The Baltic Proper subpopulation of *Phocoena phocoena*, of which fewer than 500 survive (SAMBAH 2016, North Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research 2019, Siebert et al. 2020, Owen et al. 2021, Amundin et al. 2022), and the vaquita (*Phocoena sinus*), of which fewer than 10 survive (Jaramillo-Legorreta et al. 2019, Robinson et al. 2022), are arguably in a more perilous state. However, it seems clear that urgent management action is needed to secure the long-term viability of the Iberian porpoise population. The present review encompasses the taxonomic status of the Iberian porpoise, its biology and ecology (highlighting differences from other harbour porpoise populations where applicable), evidence about the status of the population, the various natural and anthropogenic threats it faces – notably mortality due to fishery bycatch, the legal framework under which it is protected and current conservation management. It also aims to identify knowledge gaps and actions necessary to protect this vulnerable population.

Taxonomy, genetic status and phylogeography of the Iberian population

Porpoises from the Iberian Peninsula reach larger sizes than those from further north within Europe (e.g. Smeenk et al. 1992, Donovan & Bjørge 1995, Sequeira 1996, López Fernández 2003, López-Fernández & Martínez-Cedeira 2011). As genetic evidence was amassed, it became clear that Iberian porpoises form a morphologically and genetically distinct, largely isolated, population (Fontaine et al. 2007, 2010, 2014, Celemín et al. 2023) closely related to the population in Northwest Africa (Fontaine et al. 2014). Fontaine et al. (2014; see also the review by Fontaine 2016) proposed that the Iberian and Northwest African porpoise populations together represent a distinct ecotype adapted to upwelling systems. Considering their phylogenetic divergence from the subspecies described in the North Atlantic (*P. p. phocoena*) and Black Sea (*P. p. relicta*), their allopatric distribution, and their morphological and ecological distinctiveness, it was proposed to raise this distinct ecotype as a separate subspecies with the name *P. p. meridionalis* (Fontaine et al. 2014, Fontaine 2016) (Figures 2 and 3). Ben Chehida et al. (2021) and Olsen et al. (2022) identified a fourth porpoise lineage from West Greenland, an area where an offshore ecotype is known to exist (Nielsen et al. 2018). They argued that this fourth lineage probably split off at the same time the other lineages emerged and speculated that it emerged in oceanic waters around the Azores. However, there are very few porpoise records from Macaronesia, and there have been no studies of their genetic makeup.

Within the Iberian Peninsula, porpoises from Spain and Portugal are genetically similar and form part of a single population; no differences in mitochondrial and microsatellite genetic diversity, or in genetic ancestry, were detected between samples from the two countries (Fontaine et al. 2007, 2010, 2014, Llavona Vallina 2018, Ben Chehida et al. 2021, 2023). It should be noted that samples usually come from stranded animals, and given the prevailing currents, it is possible that some porpoises stranded along the northwest coast of Spain had died in Portuguese waters. Genetic diversity of the Iberian population was lower than that of porpoises from the Bay of Biscay, although comparable with that in Mauritania and the Black Sea. Despite similar genetic diversity, the abundance of the Iberian porpoise is around 30 times lower than that of the Black Sea population (see ACCOBAMS 2021 and section ‘Distribution and abundance’ of this chapter). Ben Chehida et al. (2023) reported that a sharp decline in the genetic diversity of Iberian porpoises occurred between 1990 and 2015, caused by genetic drift and/or a decline in the effective population size, possibly driven by environmental stochasticity, prey depletion or bycatch mortality. Low genetic diversity was also identified in Iberian porpoises by Celemín et al. (2023), in addition to high levels of inbreeding and a low effective population size.

Phylogeographic analyses by Fontaine et al. (2014) suggested that the upwelling ecotype of harbour porpoise present in the Iberian Peninsula and Mauritania descended from a now extinct paleo-population living in the Western Mediterranean Sea during the last glacial maximum (~20,000 years before present) (Figure 3). Porpoises likely entered the Mediterranean Sea from the Northeast Atlantic and split from the Atlantic populations within the past ~30,000 years. Porpoises subsequently disappeared from the Mediterranean during the postglacial warming period, but the Western and Eastern Mediterranean lineages, respectively, gave rise to the ‘upwelling’ and Black Sea groups, around 15,000 years ago, with the former giving rise to the Iberian and North West African groups (Figure 3).

Population genetic analyses to date suggest that porpoises from the Iberian population mix asymmetrically northwards with those from the Bay of Biscay and the Celtic Sea, with individuals migrating away from the Iberian unit but not into it (Alfonsi et al. 2012, Fontaine et al. 2014, 2017, Ben Chehida et al. 2021). Porpoises found stranded on the northern side of the Bay of Biscay and the Celtic Sea clearly displayed an admixed genetic ancestry, defining a geographically localised tension (or hybrid) zone where most porpoises not only show mixed genetic ancestry between Iberian porpoises and those further north but also exhibit the larger body size typical of the Iberian porpoises (Fontaine et al. 2017). Fontaine et al. (2010, 2014; reviewed in Fontaine 2016) estimated that porpoises from Iberian waters and those further north came back into contact and established this hybrid zone ~300 years ago. Mitochondrial evidence also suggested that individuals from the Iberian population interbreed with the population in Northwest African waters (or with an unknown population in between) (Fontaine et al. 2014, Ben Chehida et al. 2023). Northwest African porpoises are most closely related to Iberian animals, both being descended from the extinct western Mediterranean population and diverging from each other around 3000 years ago. Overall, it seems clear that the porpoises of the Iberian Peninsula should be treated as a distinct unit.

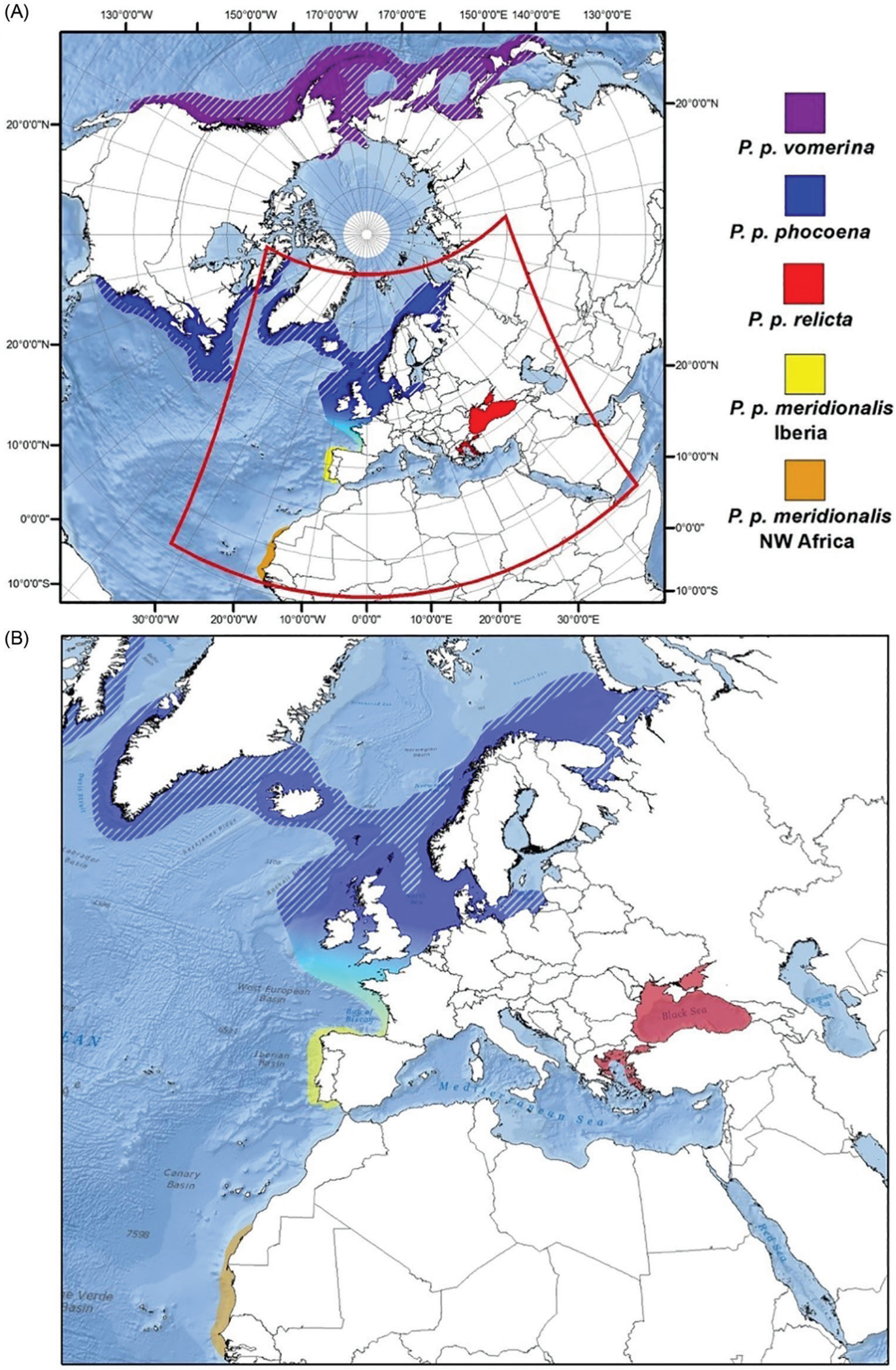

Figure 2 Distribution of the harbour porpoise (*Phocoena phocoena*), showing ranges for the different subspecies, including the proposed *P. p. meridionalis*, and genetically differentiated populations. (A) The global circumpolar distribution of the harbour porpoise. Polygon (in red) indicates the enlarged view displayed in (b). (B) The Northeast Atlantic distribution of the species with the different subspecies and genetically distinct

(*Continued*)

(Continued)

populations bordering the Mediterranean Sea. The four subspecies are displayed with different colours. The known and possible species distribution is shown with plain and hashed surfaces, respectively. A coloured gradient (yellow to dark blue) in the south of the British Isles and Bay of Biscay shows the approximate geographic distribution of the admixture zone between the Iberian population of *P. p. meridionalis* and the population of *P. p. phocoena* north of the Bay of Biscay. The Figure was prepared using ESRI ArcGIS v.10.3 and is based on maps in Gaskin (1984), IWC (1996) and Read (1999), updated with recent observations from the Black Sea and the northern Aegean Sea. (Reprinted from *Advances in Marine Biology*, Vol 75, Michael C. Fontaine, Harbour Porpoises, *Phocoena phocoena*, in the Mediterranean Sea and Adjacent Regions: Biogeographic Relicts of the Last Glacial Period, Page 335, Copyright (2016), with permission from Elsevier.)

Additional evidence about population structure and movements in cetaceans can be derived from ecological tracers such as stable isotope signatures, contaminant concentrations and parasites. Although differences in stable isotope profiles have been used to infer the existence of distinct groups of harbour porpoises within an area (e.g. Jansen et al. 2012) and the absence of differences has been used to infer movements of porpoises between areas (e.g. Angerbjörn et al. 2006), to date such studies have not included Iberian animals. Studies of concentrations of metals and/or organic contaminants in porpoise tissues can provide similar kinds of insights (e.g. Das et al. 2004, Pierce et al. 2008) but again have little to say about the distinctiveness of Iberian porpoise. We revisit this topic in section 'Threats to the Iberian harbour porpoise - Pollutants and other harmful substances', where the role of pollutants as a threat to Iberian porpoise is discussed.

Parasites, including larval nematodes, are routinely used in stock identification for fish (e.g. MacKenzie et al. 2008, MacKenzie & Hemmingsen 2015). In principle, the same approach is applicable to cetaceans, which are the final hosts of *Anisakis* spp. (MacKenzie 2002). Nematodes found in the digestive tract of cetaceans derive from their prey and hence may be useful to identify "ecological stocks" of cetaceans with different habitats and feeding habits, perhaps also true (reproductively isolated) stocks. Gomes et al. (2021) recently found a strong correlation between the proportions of mature nematodes belonging to the sibling species *Anisakis pegreffii* and *Anisakis simplex* (*s.s.*) and stock identity of minke whales (*Balaenoptera acutorostrata*) in the waters around Japan. Results of molecular analysis of *Anisakis* parasites from a small sample of harbour porpoises provided a preliminary indication that parasitological analysis could be used as a complementary method supporting the proposed *P. phocoena meridionalis* subspecies (Cipriani et al. 2022). In one porpoise from Scotland, all the *Anisakis* identified were *A. simplex* (*s.s.*), while three porpoises from Galicia all had mixed infections, with 19% overall of the adult *Anisakis* worms being *A. pegreffi*. This result was consistent with the known distribution of the different *Anisakis* species in the Northeast Atlantic, with *A. pegreffi* being absent from northern waters, while the two species occur sympatrically in fish inhabiting Atlantic waters of the Iberian Peninsula (Abollo et al. 2003, Levsen et al. 2018). Both *Anisakis* species were also present in similar proportions in blue whiting *Micromesistius poutassou* (Roca-Geronès et al. 2020), which is an important prey species for porpoises in the area (Hernandez-Gonzalez et al. 2024).

For the assessment of the status of harbour porpoise in European Atlantic waters in relation to various indicators under the EU Marine Strategy Framework Directive (MSFD), ICES (2014a,b) proposed the use of five Assessment Units, with the Iberia Peninsula being treated as a separate Assessment Unit (Figure 4A). It should be noted that these Units did not include the Baltic Proper sub-population. The OSPAR Commission (2023) subsequently proposed nine Assessment Units for the MSFD bycatch indicator (M6), extending the Iberian porpoise AU into offshore waters, changing the AUs west of the UK and including additional AUs in the north (but still not including the Baltic Sea proper sub-population) (Figure 4B).

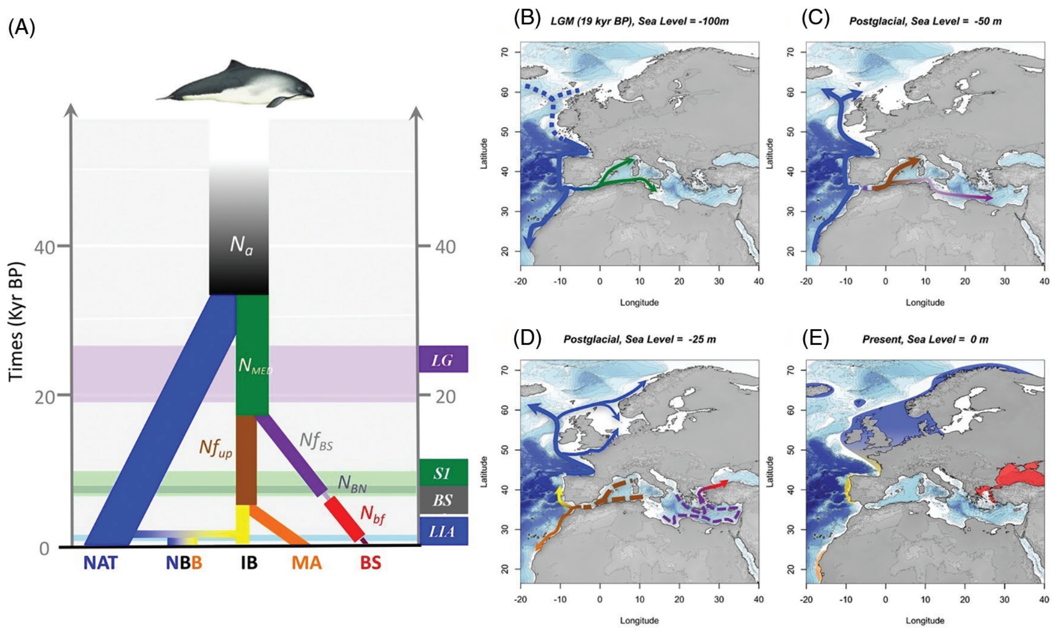

Figure 3 Evolutionary history of the harbour porpoise (*Phocoena phocoena*) as determined from genetic analyses. (A) The evolutionary history of the differentiated groups with a schematic population tree. Each group is shown at the bottom with a colour coding following Figure 1 (BS, population from the Black Sea *P. p. relicta*; IB, Iberian population of the southern ecotype, the proposed *P. p. meridionalis*; MA, Mauritanian–Northwest African population of the southern ecotype; NAT, Northeast Atlantic ecotype (*P. p. phocoena*) inhabiting the European continental shelf north of the Bay of Biscay; NBB, admixed population between IB and NAT in the northern side of the Bay of Biscay). Each group coalesces backwards in time (upward) as indicated by the *Y*-axis, which provides the timeline in thousands of years before present (kyr BP). Changes in population size are depicted by changing line width. Major environmental changes related to the demographic history of harbour porpoises are also plotted: Last Glacial Maximum (LGM period, ca. 23–19 kyr BP) (Clark et al. 2009), Mediterranean Sapropel S1 period (ca. 9.5–6.5 kyr BP) (Spötl et al. 2010, Roberts et al. 2011), flooding of the Black Sea (BS, ca. 8.4–9.4 kyr BP) (Major et al. 2006, Giosan et al. 2009) and Little Ice Age (LIA, ca. 250–700 yr BP) (Osborn & Briffa 2006). (B–E) The biogeographic landscape of porpoise evolutionary history at four time-steps determined by historical changes in sea level: (B) During the LGM (glacial sea level 100 m lower than today), porpoises from the Atlantic likely colonised the Mediterranean Sea, resulting in a divergent group that formed the ancestral Mediterranean population(s) (N_{MED} in A); (C) During the postglacial Holocene warming and the sea level rise, these ancestral populations in the Mediterranean Sea split into eastern (purple, Nf_{BS} in A) and western (brown, Nf_{UP}) lineages, from which descended the porpoise of the Black Sea and the two populations of the upwelling waters of Iberia and Mauritania–Northwest Africa; (D) Mediterranean conditions became unsuitable for the harbour porpoise at the end of the African Humid Period and Mediterranean Sapropel episodes. Porpoises were thus forced out of the Mediterranean, taking refuge where conditions were still suitable for the species: the Eastern Mediterranean lineage moved into the Black Sea, reconnected to the Mediterranean ca. 8.4 kyr BP, and the Western Mediterranean lineage moved back into the Atlantic waters of Iberia and Northwest Africa; (E) The present distribution. Maps were drawn using MARMAP v0.9.5 package (Pante & Simon-Bouhet 2013) for R (R Core Team 2016) using the ETOPO1 dataset available on the United States National Geophysical Data Center (Amante & Eakins 2009). Sea level was adjusted to account for the historical variation during the LGM and post-LGM period. Panel (A) was based on Fontaine et al. (2012, 2014). (Reprinted from *Advances in Marine Biology*, Vol 75, Michael C. Fontaine, Harbour Porpoises, *Phocoena phocoena*, in the Mediterranean Sea and Adjacent Regions: Biogeographic Relicts of the Last Glacial Period, Page 341, Copyright (2016), with permission from Elsevier.)

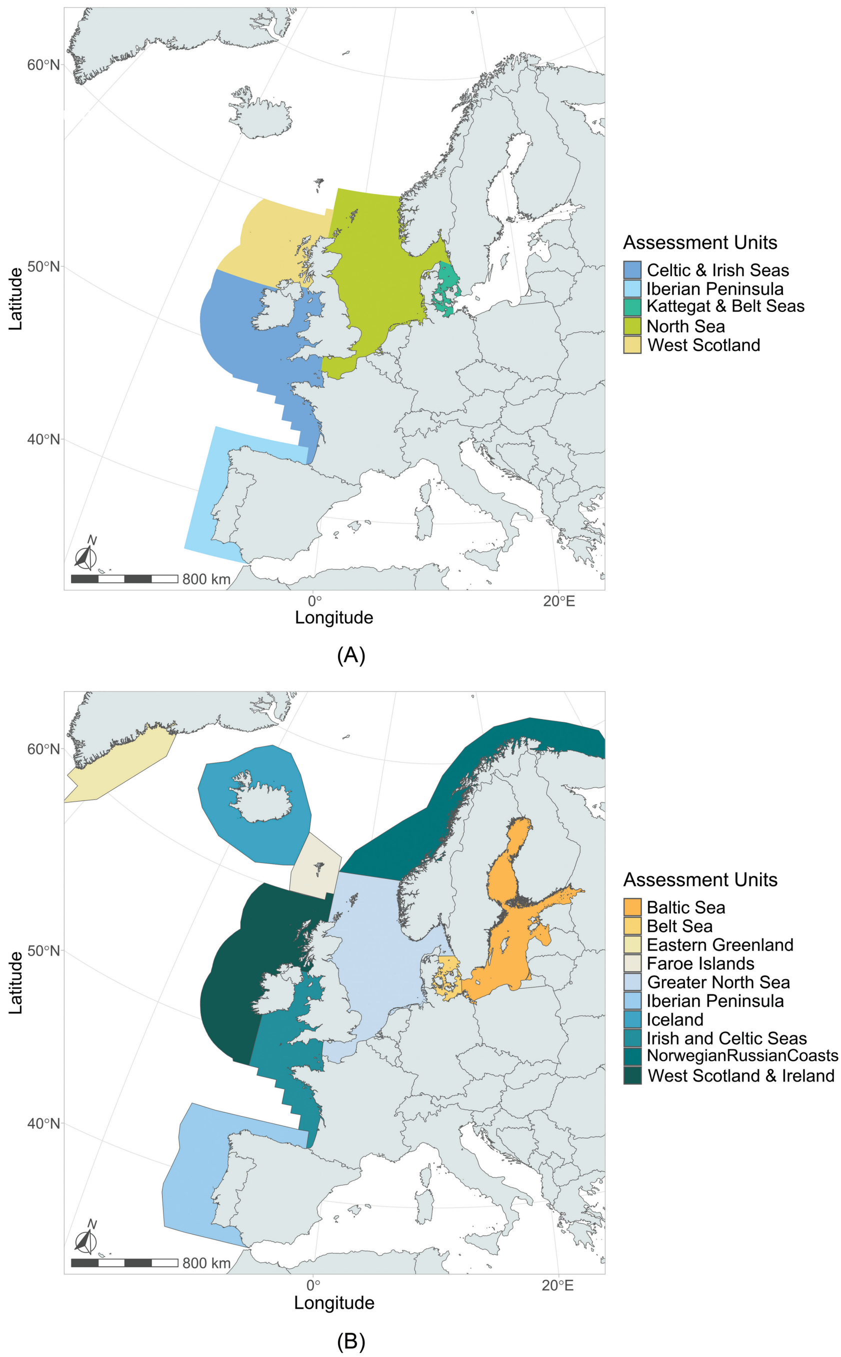

Figure 4 (A) ICES Assessment Units for harbour porpoise. (Recreated from information in Figure 1.6.6.1.2 in ICES (2014a) and Figure 1 in Appendix 1 of ICES (2014b).) (B) OSPAR Commission Assessment Units for M6 assessment units for harbour porpoise, based on Figure c in Geelhoed et al. (2022) and Figure 1 in OSPAR Commission (2023), which in turn were based on a proposal by North Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research (2019), plus the Baltic Sea proper Assessment Unit used by HELCOM (2023). The figure was reconstructed using the geopackage https://github.com/osparcomm/Abundance-and-Distribution-of-Cetaceans.

Distribution and abundance

Globally, the harbour porpoise is generally considered to be a coastal species preferring continental shelf waters (e.g. Evans 1987) although there is evidence from satellite tagged animals of movements into deep oceanic waters off Greenland (which has a very narrow continental shelf), with porpoises there diving to depths of up to 410 m (Nielsen et al. 2018, Olsen et al. 2022). As mentioned in the previous section, Ben Chehida et al. (2021) and Olsen et al. (2022) referred to these animals as representing an offshore ecotype and detected a genetically distinct lineage in the region.

The Iberian harbour porpoise population inhabits the cold-water upwelling zone along the Atlantic coasts of Spain and Portugal, from the south Biscay coast to (at least) the Algarve coast of Portugal, bordering the Gulf of Cádiz (Sequeira 1996, Castro 2010). Recent occurrence records are most numerous in the Galicia region of Spain and in northern and central Portugal (e.g., Donovan & Bjørge 1995, Sequeira 1996, Fontaine 2016, Read 2016, Hammond et al. 2017, Torres-Pereira et al. 2022, Gilles et al. 2023). Porpoises are largely absent from the Mediterranean Sea (Frantzis et al. 2001), and there were no sightings of porpoises in the Mediterranean Sea during the Survey Initiative of the Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic Area (ACCOBAMS) in the Mediterranean and Black Sea in 2018 (ACCOBAMS Survey Initiative 2018). However, there is evidence of occasional incursions from Iberian Atlantic waters into the contiguous Mediterranean Sea (e.g. Lens 1997, Cabezón et al. 2004, Sociedad Española de Cetáceos 2006, Anon. 2021). Cabezón et al. (2004) referred to a porpoise stranded in Andalucía (Spanish Mediterranean coast), and Lens (1997) mentioned two records from the Strait of Gibraltar in the early 1980s. Porpoises are also known from the northern Aegean Sea where, for example, a number of sightings and acoustic detections were recorded in both Greek and Turkish waters during a survey in 2013 (Cucknell et al. 2016). Genetic analyses of stranded porpoises from this region indicated that these animals belong to the same genetic pool as Black Sea porpoises (Fontaine et al. 2012).

There are also isolated records from the Azores (Barreiros et al. 2006) and Canary Islands (Díaz-Delgado et al. 2018), although the affiliation of these animals cannot be confirmed, and at least in the Canary Islands, it is more likely that such animals derive from the northwest African population. It is worth noting that the continental shelf of the Iberian Peninsula is quite narrow but to date almost all sightings are from shelf waters, despite several surveys extending into offshore waters. However, the probability of detecting porpoises during visual surveys in oceanic waters, in all but the calmest conditions, is modest.

Large-scale dedicated abundance surveys

The only surveys that have covered almost the whole Atlantic Iberian coast were SCANS-II in 2005 (with the CODA survey in 2007 covering offshore waters), SCANS-III in 2016 and SCANS-IV in 2022, although these surveys did not extend into the interior waters of the Galician Rías. All three surveys took place in summer (June–July). The 2005 SCANS-II survey (Figure 5A and C) covered shelf waters of the Iberian Peninsula and the southern and central Bay of Biscay (Block W) and produced abundance and density estimates for this area of 2357 animals (CV = 0.92) and 0.017 animals km^{-2} (CV = 0.92), respectively (Hammond et al. 2013). These estimates were revised by Hammond et al. (2021) to 2880 animals (CV = 0.72) and 0.021 (CV = 0.72) animals km^{-2}, respectively. A complementary survey of adjacent offshore waters (CODA) was carried out in 2007, comprising shipboard visual and acoustic surveys. No porpoises were detected visually and only one acoustically (off southwest of Britain) (Hammond et al. 2009). For the 2016 SCANS-III survey (Figure 5B and D), the survey area was amended to correspond to the Iberian Peninsula Management Unit (IPMU) as proposed by ASCOBANS, the Baltic Marine Environment Protection Commission (better known as the Helsinki Commission or HELCOM) and ICES (Evans & Teilmann 2009, ICES 2009, 2013b, 2014a), which in turn reflected the recent genetics studies on harbour porpoises. The resulting survey

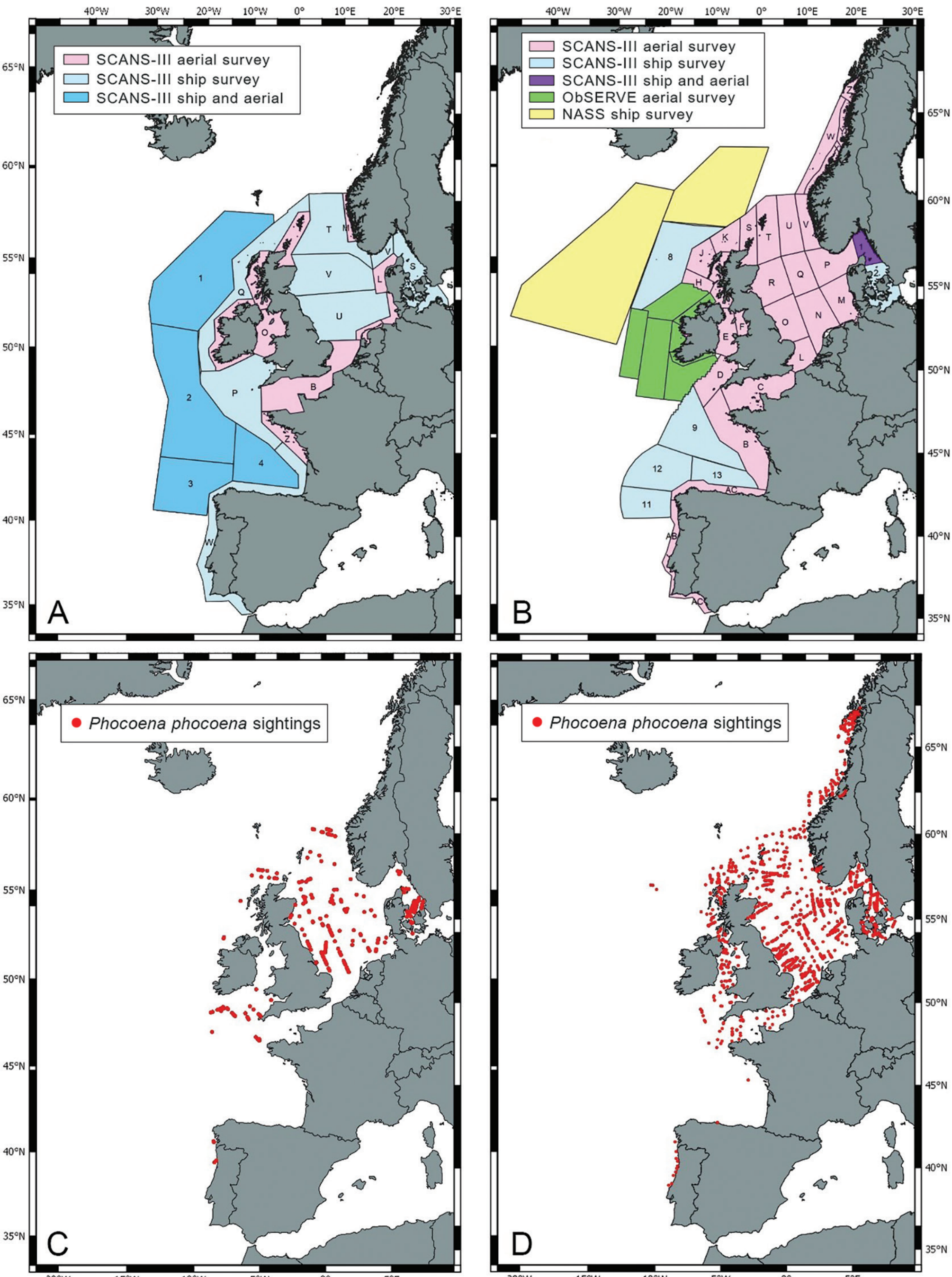

Figure 5 Porpoise sightings from the SCANS II and III surveys: (A) SCANS-II (2005) and CODA (2007) survey blocks, (B) SCANS-III (2016) and related survey blocks, (C) porpoise sightings from SCANS II, (D) porpoise sightings from SCANS III survey. For harbour porpoise, the usable effort comprised aerial surveys carried out in good and moderate conditions and ship surveys undertaken in Beaufort 0–2. (Panels A and B are adapted from Lacey et al. (2022), panel C is based on SCANS-II (2008), recreated using SCANS-II data from OBIS. Panel D is modified from Hammond et al. (2021). (Figures reproduced with permission from Phil Hammond.)

Table 1 Harbour Porpoise Abundance and Density (animals km^{-2}) Estimates from the Iberian Peninsula Block A of the SCANS-III Survey in 2016 (Hammond et al. 2017)

Block	Geographic Region	Abundance	Density	CV	CL Low	CL High
AA	Strait of Gibraltar to Cabo de São Vicente	0	0	0	0	0
AB	Cabo de São Vicente to Cape Finisterre	2715	0.102	0.308	1350	4737
AC	Cape Finisterre to Bayonne (France), including the southern Bay of Biscay	183	0.005	1.020	0	669

CV is the coefficient of variation of abundance and density. CL low and CL high are the estimated lower and upper 95% confidence limits of abundance.

block (Block A) was further divided into three sub-blocks spanning the Atlantic and Bay of Biscay coasts of Portugal and Spain. SCANS-III generated an abundance estimate of 2715 individuals (CV = 0.31) for sub-block AB, from Cabo de São Vicente in Portugal northwards to Cape Finisterre in Galicia, which encompasses the core range of the Iberian population (Table 1). No porpoises were seen in sub-block AA from the Strait of Gibraltar to Cabo de São Vicente in Portugal, and few were seen in block AC from Cape Finisterre to Bayonne (France) (Figure 5D, Table 1). The combined abundance estimate for the IPMU was 2898 animals (CV = 0.32) (Hammond et al. 2017, 2021).

The porpoise densities recorded for the Iberian Peninsula SCANS blocks during the 2005 and 2016 surveys were among the lowest over the entire European continental shelf. The abundance estimates for the IPMU were also very similar, 2880 (CV = 0.72) and 2900 (CV = 0.32), respectively, once the former had been corrected for likely negative bias (Hammond et al. 2017, 2021). The similarity of the two abundance estimates suggests that there was no upward or downward trend over this period, although the estimates of porpoise abundance for the Iberian Peninsula are based on a small number of sightings, and such trends could be accommodated within the 95% confidence intervals. It is worth noting that almost all the porpoise sightings along the Iberian coast during the SCANS-III survey were between Viana do Castelo (near the Spanish border) and Peniche (north of Lisbon) along the west coast of Portugal (Figure 5D). SCANS-IV took place in summer 2022. The survey area off the Iberian Peninsula extended into offshore waters, although all porpoise sightings were in inshore waters, and a majority of them were along the western coast of Galicia. The resulting abundance estimate for the Iberian AU was somewhat higher (4043, with a CV of 0.35% and 95% confidence limits of 1842–7309) (Gilles et al. 2023). Note that all calculations of the potential impact of fishery bycatch in section 'Threats to the Iberian harbour porpoise' were based on the previous population estimate of approximately 2900 animals.

National and regional surveys plus opportunistic sightings

Numerous national and regional surveys have recorded cetacean distribution along the Spanish and Portuguese Atlantic coasts (Figure 6) but only a few have generated abundance estimates. The relatively low numbers of porpoise sightings resulted in wide confidence limits around the abundance estimates, and estimates from Portuguese surveys also varied markedly between years (ICES 2014a). The results seem to be consistent with the conclusion from the SCANS surveys that porpoise density in Iberian waters is lower than in most European Atlantic shelf waters. A low sighting rate is also reported for the adjacent porpoise population in Northwest Africa. It was not until 2007 that Boisseau et al. (2007) reported the first sightings of live harbour porpoises (three sightings in Agadir Bay (Morocco) during almost 2000 km of effort). Consistent with the results of large-scale surveys (e.g. Figure 5), surveys carried out both in Portugal and northern Spain suggest that the majority of the Iberian porpoise population inhabits Portuguese waters.

The general picture that emerges from sightings in Spanish waters is of small numbers of records along most of the coast and a few hotspots in Galicia, notably over the continental shelf outside the Ría of Arousa, although the high number of sightings here reflects high survey effort,

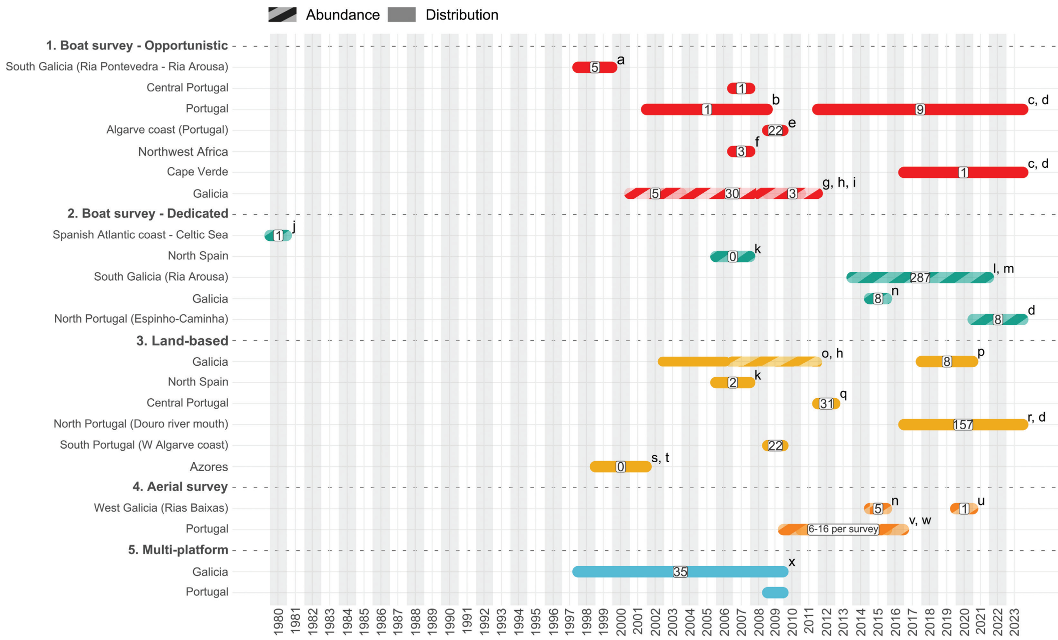

Figure 6 Graphical representation of the different types of surveys, regional and national, carried out in Iberian Peninsula waters between 1980 and 2023 that have collected harbour porpoise sightings. The bars represent the years during which surveys were carried out, plain coloured bars represent surveys that allow determination of porpoise distribution and bars with diagonal line pattern represent surveys from which abundance estimates can be inferred. The number inside each white box represents the number of sightings of harbour porpoise registered (when available) and the letters at the end of the bars indicate the source of the data (when available), as cited in the main text, and detailed below: (A) López et al. 2004; (B) Brito et al. 2009; (C) Correia et al. 2019; (D) A. Gil, pers. comm.; (E) Castro 2010; (F) Boisseau et al. 2007; (G) Spyrakos et al. 2011; (H) Llavona Vallina 2018; (I) López et al. 2011; (J) Aguilar et al. 1983; (K) López et al. 2013; (L) Díaz López and Methion 2018; (M) Díaz López et al. 2022; (N) Martínez-Cedeira et al. 2015; (O) Pierce et al. 2010; (P) López, pers. comm.; (Q) Pereira 2015; (R) Gil et al. 2019; (S) Silva et al. 2011; (T) Barreiros et al. 2006; (U) Martínez-Cedeira et al. 2021; (V) Vingada et al. 2011; (W) Torres-Pereira et al. 2022; (X) Fernández et al. 2013.

since the Bottlenose Dolphin Research Institute (BDRI) is based in the area, coupled with an almost complete absence in deeper offshore waters. In Portugal, there have been more sightings, although with considerable variation between years and between locations.

Spain

Based on the data collected between 2003 and 2011 from multiple sources, López et al. (2013) produced an abundance estimate for harbour porpoises in the Spanish Galician and Bay of Biscay waters of 683 animals (CV = 0.63, 95% CI: 345–951, N = 40), with a density estimate of 0.0008 animals per km^2. This estimate did not account for availability, perception or responsive movement biases and may therefore have been negatively biased. Summing the Portuguese and northern Spanish estimates, the total estimated abundance for Atlantic waters of the Iberian Peninsula excluding the Gulf of Cadiz (2254 + 683 = 2937 animals) is similar to the SCANS II and III abundance estimates. López et al. (2012) provided a separate abundance estimate of 386 (CV = 0.71) porpoises in Galician waters (see also CEMMA 2018).

The French SAMM (Suivi Aérien de la Mégafaune Marine, Aerial Census of Marine Megafauna) summer and winter aerial surveys extended to the eastern part of the North coast of Spain. All the porpoise sightings in the southern part of the Bay of Biscay were along the French coast. It was apparent that sightings of this species extended further south in winter (to within 50 km of the Spanish border), mainly along the coast, while in summer occurrences extended further offshore but no porpoises were seen further south than Royan (Lambert et al. 2017 (in their Appendix S4)). This seasonal pattern thus probably relates mainly to animals from the main European population. The SAMM II winter survey involved a higher survey effort (more closely spaced transect lines), and there were more on-effort sightings on porpoises, again including some very close to the Spanish border (Blanchard et al. 2021).

Lens (1997) noted that there were few records of porpoise sightings in the Iberian Peninsula, mostly comprising opportunistic sightings along the west and north coasts. Cabrera (1914) referred to porpoises being present around the Strait of Gibraltar, and there were two further records from this area in 1981, comprising one stranding in Málaga (Rey & Cendrero 1982) and one sighting (Duguy & Desportes 1983). The Ballena surveys, which took place annually in the first half of the 1980s, extended from the Spanish Atlantic coast to the Celtic Sea and focused on (the then still exploited) fin whale *Balaenoptera physalus*. The Ballena 1 survey also yielded a single porpoise sighting, well off the continental shelf, 140 miles west of Lisbon (Aguilar et al. 1983), while surveys 2, 3 and 4 reported small cetaceans but no porpoises were seen (Sanpera et al. 1984, 1985, Sanpera & Jover 1986). McBrearty et al. (1986) mentioned sightings of porpoises along the Spanish and Portuguese Atlantic coasts from British Royal Navy and Merchant Navy vessels during 1978–1982.

During 4 years of shore-based monitoring of the Galician coast (September 2003–October 2007) by CEMMA, involving monthly visits to at least 30 sites, porpoises sightings comprised 10.4% of all coastal cetacean sightings (3.4% of the estimated total number of individuals) and were distributed all along the Galician coast, with the highest sighting frequencies recorded from Faro Punta Roncadoira on the north coast of Galicia, Faro Cabo Vilán near Cabo Fisterra (the westernmost point of Galicia) and A Guarda (on the border with Portugal) (Pierce et al. 2010). Llavona Vallina (2018) repeated the analysis using data up to the end of 2011: over the whole period, porpoise sightings comprised 12.3% of all cetacean sightings. Average annual encounter rates showed a slight overall upward trend (Figure 7). The relative frequency of porpoise sightings was almost unchanged when the study concluded at the end of 2013 (12.1%; A. López, Pers. Comm.)

Combining all boat survey data for Galicia during 1998–2009, there were 35 porpoise sightings, all recorded between Cabo Fisterra and the Portuguese border, suggesting that the south-west region of Galicia is of particular importance for this species (Fernández et al. 2013). Boat-based surveys in Galician waters during 2003–2010 ($N = 111$), using a variety of vessels (including fishing vessels), resulted in a total of 30 porpoise sightings during approximately 844 hours of survey effort, mostly in shelf waters up to 200 m deep (Llavona Vallina 2018). Several of the surveys included in the above-mentioned totals recorded few porpoises, including those using observers onboard fishing vessels in Galician coastal waters during 1998–1999 (López et al. 2004) and 2001–2003 (Spyrakos et al. 2011). The 1998–1999 surveys covered approximately 20,000 km^2 and recorded five porpoise sightings in shallow waters adjacent to the Rías of Pontevedra and Arousa in southern Galicia (López et al. 2004). Several boat-based surveys recorded no porpoises, including surveys along the entire northern Spanish coast in 2006 and 2007 (López et al. 2013), also dedicated surveys of the Galician Bank (180 km off the Galician coast in water depths of 650–1500 m in 2006–2007 and 2009–2011) and the Avilés Canyon (north of the coast of Asturias in waters with depths between 140 m and 4700 m in 2009–2011) (Llavona et al. 2011).

The NGO CEMMA has continued carrying out opportunistic and dedicated surveys. Observations onboard fishing vessels in 2009–2011 in Northwest Spanish shelf waters resulted in three harbour porpoise sightings among 80 cetacean sightings (López et al. 2011, CEMMA 2012).

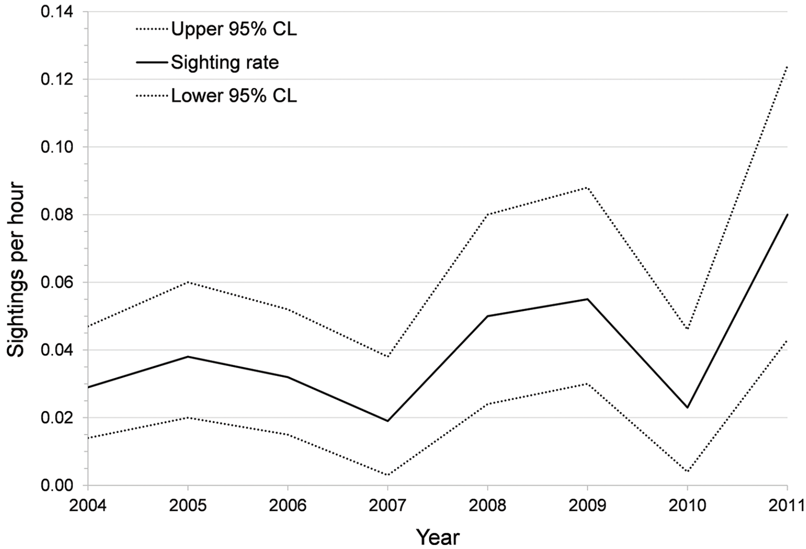

Figure 7 Annual average encounter rates (porpoise sightings per hour) from land-based surveys along the Galician (Northwest Spain) coast between January 2004 and December 2011, across all observation points, with 95% confidence limits (CL, dashed lines). Data collection took place monthly from at least 30 sites. (Adapted from Llavona Vallina 2018.)

Dedicated boat surveys in Galicia in September 2015, which covered an area of nearly 4000 km^2 from the coast (including interior waters of the Rías) to the 200 m isobath, registered eight harbour porpoise sightings among 133 cetacean sightings (Martínez-Cedeira et al. 2015). Aerial surveys carried out in September in 2015 and 2020 in the Rías Baixas (southwest Galicia), again extending to the 200 m isobath, registered five sightings and one sighting of harbour porpoise, respectively, out of 111 and 93 total cetacean sightings (Martínez-Cedeira et al. 2015, 2021). Surveys of coastal waters from the Ría de Vigo to Fisterra during 2018–2020 resulted in eight porpoise sightings among 216 cetacean sightings (A. López, Pers. Comm.).

Boat-based surveys carried out by the Bottlenose Dolphin Research Institute (BDRI) between 2014 and 2021 highlighted that the continental shelf of southern Galicia is a hotspot for harbour porpoises (Díaz López & Methion 2018, Díaz López et al. 2022). During 195 daily dedicated surveys extending to the 1400 m isobath, 287 groups of harbour porpoise were recorded throughout the southern Galician continental shelf, including the Ría de Arousa (Figure 8). All harbour porpoises were observed within the Ría de Arousa and outside the Ria over the continental shelf, in waters with a minimum depth of 2 m and a maximum depth of 231 m (mean depth = 84 m, SE = 2.8 m). The encounter rate inside the Ría de Arousa (0.05 sightings per hour from 1055 hours of observation) was markedly lower than the sighting rate from continental shelf waters outside the Ria (0.42 sighting per hour from 562 hours of observation). A small proportion of survey effort (13 hours) took place beyond the shelf edge but there were no porpoise sightings in these offshore waters.

Along the southern Galician continental shelf, porpoise group size ranged from single individuals to groups of 46 (mean = 4.8 individuals per group, SE = 0.3). The sighting rates were notably higher in summer and autumn than in winter or spring, which may be related to the calving period (see section 'Life history') as well as to high seasonal productivity due to coastal upwelling (Díaz López & Methion 2018) (see Table 2).

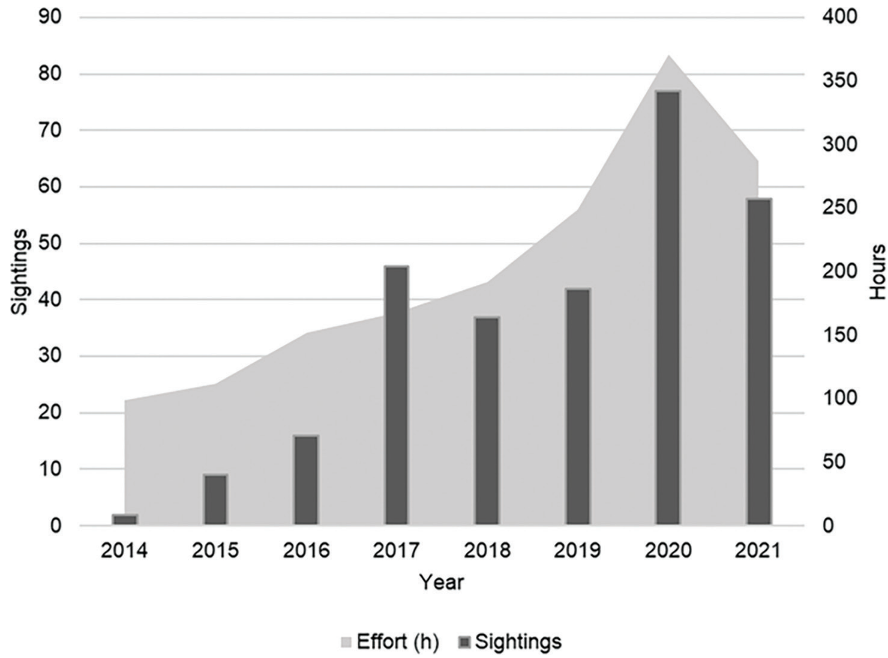

Figure 8 Annual numbers of sightings of groups of porpoises (histogram bars) and survey effort in hours (shaded area) from BDRI boat-based surveys along the Southern Galician coast (Northwest Spain) between April 2014 and November 2021. Data collection took place monthly and year-round. (Adapted from Díaz López et al. 2022.)

Table 2 Seasonal Distribution of Harbour Porpoise Sightings and Group Size

Season	Effort (hours)	No. of Sightings	SPUE (sightings/hour)	Mean Group Size±s.e	No of Individuals	SPUE (adults/hour)
Winter	235	11	0.05	4.1±1.9	45	0.19
Spring	414	42	0.10	3.3±0.4	140	0.34
Summer	736	176	0.24	5.1±0.4	889	1.21
Autumn	245	58	0.24	5.2±0.7	309	1.26
Whole year	1630	287	0.18	5.5±0.3	1421	0.87

Source: Adapted from Díaz López et al. (2022).

Porpoise sightings appear to be rare off southern Spain in the Gulf of Cádiz, with only seven sightings registered in the area (Sociedad Española de Cetáceos 2006), although strandings data confirm the regular presence of porpoises in the Gulf of Cádiz (see section 'Distribution and abundance - Strandings'). Porpoises are generally absent from the Strait of Gibraltar and the western Mediterranean Sea (Frantzis et al. 2001), although two opportunistic sightings were recorded in 2006 in coastal waters off Málaga (Sociedad Española de Cetáceos 2006). There is evidence of increased porpoise presence in the Málaga area since 2011, with a total of over 30 sightings of porpoises from tourist vessels (Samantha Blakeman, Pers. Comm.; see also Pierce et al. 2024).

Portugal

Annual aerial surveys of Portuguese coastal waters were carried out during 2011–2015, in September or October, leading to an overall abundance estimate of 2254 (95% CI = 1287–3949) (Torres-Pereira et al. 2022). There were between 6 and 16 sightings (8–25 individuals) per survey, and the resulting annual abundance estimates ranged from 1196 animals (95% CI = 456–3135) in 2011 to 3207 animals (95% CI = 1531–6718) in 2013. The authors highlighted a sharp drop in estimated abundance between 2013 and 2014 (a 48% decrease to 1653 animals (95% CI = 717–3809)). However, marked increases were seen from 2014 to 2015 (a 30% increase to 2147 (95% CI = 923–4997)) and from 2011 to 2012 (a 150% increase to 2416 (95% CI = 1338–4363)). Considering the small numbers of sightings on which these estimates are based, these changes are unlikely to reflect real changes in abundance and may, for example, indicate changes in distribution or simply stochastic variation in the number of animals seen at the sea surface. Partial results from these surveys were previously reported by Santos et al. (2012) and ICES (2014b).

Aerial surveys between 2008 and 2011 confirmed an important area of porpoise occurrence between Porto and Peniche, especially between Figueira da Foz and Nazaré, i.e., immediately to the south of the area of peak strandings (Vingada et al. 2011). Predicted occurrence maps generated using data from aerial surveys along the Portuguese coastline during 2010–2015 suggested that there was greater year-to-year fluctuation in occurrence in southern Portugal than in the central and northern areas (Araújo et al. 2015; see also Torres-Pereira et al. 2022).

Brito et al. (2009) compiled information on opportunistic cetacean sightings and sightings during boat-based surveys off west-central Portugal during 2002–2008. One porpoise sighting was reported among 45 opportunistic sightings of cetaceans, and no porpoises were reported among a further 45 cetacean sightings during surveys. Wise et al. (2007) recorded the presence of cetaceans in the vicinity of purse seine fishing operations and mentioned a single sighting of two porpoises off Figueira da Foz (south of Porto). During 2009, 22 porpoise sightings were recorded from a whale-watching boat along the western Algarve coast of southern Portugal (Cape São Vicente to Lagos) (Castro 2010). The Cetus project has recorded cetacean sightings from cargo ships between mainland Portugal (and occasionally mainland Spain), the Macaronesian archipelagos and West Africa since 2012. Among over 4200 cetacean sightings logged between 2012 and 2022, based on approximately 166,600 km of survey effort, there were nine sightings of harbour porpoise (Correia et al. 2019, A. Gil Pers. Comm.). Eight of these nine sightings, involving 22 individuals, were over the continental shelf of the mainland coast: one off Caracavelos at the mouth of the Tagus estuary (near Lisbon), six in the vicinity of Porto and one around 50 km northwest of Fisterra on the Galician coast.

A year of shore-based monitoring (2012) from Cabo Mondego in central Portugal produced 31 porpoise sightings, comprising 103 animals (Pereira 2015). Land-based surveys at the Douro River mouth (Porto) in northern Portugal during 2017 resulted in 22 porpoise sightings, with a maximum group size of 4 animals, and included repeated sightings of a leucistic animal (Figure 9A), both alone and in a group; on several occasions, the animals were apparently feeding (Gil et al. 2019). The above-mentioned leucistic individual was spotted with a calf on 28 July 2019 (Figure 9B) but on 20 August of the same year, it was sighted alone. It was last seen in Porto in August 2020, and it was found dead near Aveiro at the end of the month (and found to have been pregnant) (A. Torres-Pereira, Pers. Comm.). There is thus strong evidence that the porpoises that occur at the mouth of the Douro River show some site fidelity and use the area as a breeding and feeding ground. These land-based surveys have continued, involving a total of over 745 hours of observation, with 157 porpoise sightings, up to July 2023. Another leucistic individual was sighted and photographed near the Berlengas islands (off Peniche) (E. Silva, Pers. Comm.) (Figure 9C). A dataset on mammal occurrences in Portugal published in 2022 included 284 opportunistic sightings of harbour porpoise along the southern Portuguese coast during 2009–2019 by Associação para a Investigação do Meio Marinho (Grilo et al. 2022).

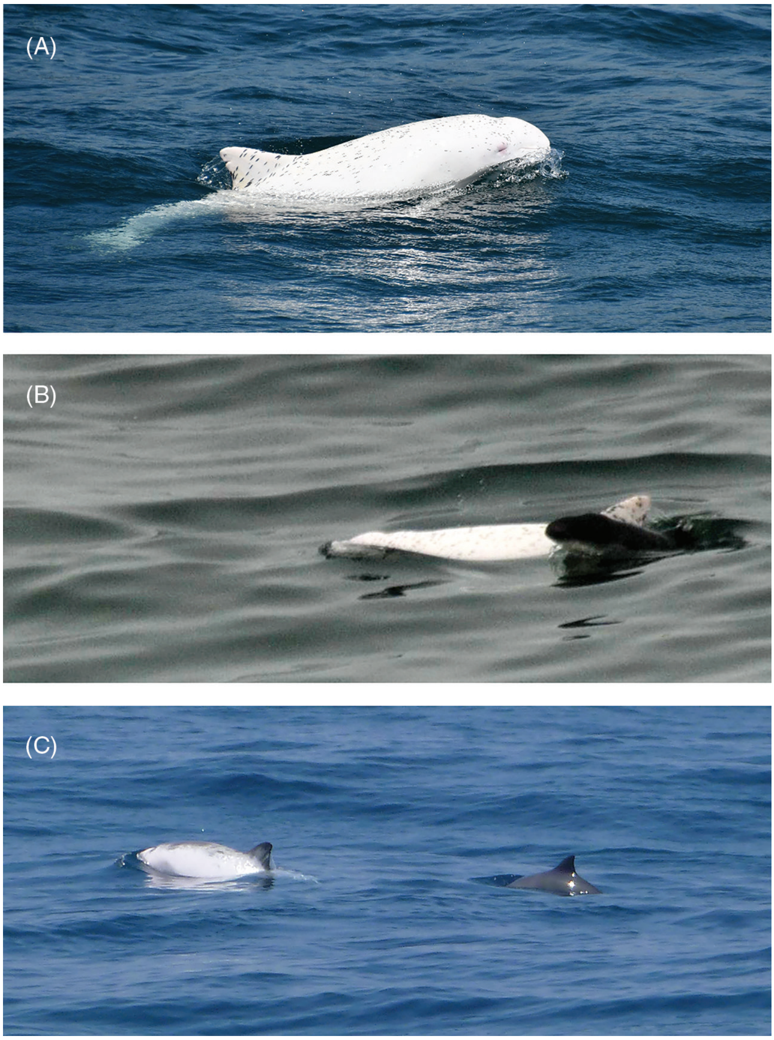

Figure 9 (A) The leucistic porpoise seen regularly around the mouth of the river Douro, Porto, Portugal (photograph by Tara Callahan, © the Cetus project), (B) Mother-calf pair of harbour porpoises sighted in the mouth of Douro river. Leucistic adult and normally-coloured calf (photography by Daniela Castelo), and (C) two porpoises, including a leucistic individual, photographed near the Berlengas Islands, off Peniche in Portugal. (Photograph by Elisabete Silva. Photographs reproduced with permission.)

There have been very few porpoise sightings reported from the Portuguese Macaronesian islands. The occurrence of harbour porpoise in Madeiran waters is based on a questionable record of a small cetacean caught in tuna fishing gear in 1905 for which no detailed description is available (Freitas et al. 2012). A total of 3564 hours of observation during 1999–2001 in the Azores, from land stations and during cruises covering the entire archipelago, coupled with use of a porpoise click detector, resulted in no visual or acoustic evidence of the presence of porpoises (Silva et al. 2001, Barreiros et al. 2006). The above-mentioned Cetus project recorded no porpoise sightings in the vicinity of Madeira or the Azores, although there was a sighting of between 20 and 40 individuals 15.7 km north of the island of Sal in the Cape Verde archipelago in waters of 2150 m depth. These were presumably animals from the Northwest African population (Correia et al. 2019, A. Gil, Pers. Comm.). Correia (2020) commented that the species is difficult to spot from a high observation platform like a cargo ship, especially in oceanic waters. Reiner et al. (1996) compiled published sightings records and their own sightings during the early 1990s for the Cape Verde Islands reporting no sightings of porpoises (although there were published records for this species in Mauritania and Senegal).

Strandings

The distribution of reported cetacean strandings in space and time will reflect the distribution of deaths at sea (presumably closely related to the distribution of living animals), as well as the prevailing winds and currents, carcass integrity and buoyancy, the presence of reporting networks and the distribution of search effort along the coast. Carcass drift models have been developed for the French coast in the Bay of Biscay (Peltier et al. 2012), in principle allowing the origin of stranded cetaceans to be inferred but this approach is not yet routinely available for the Iberian Peninsula. In Galicia, there is evidence that some carcasses may have originated in Portuguese waters, reflecting the prevalence of southwesterly winds (Saavedra et al. 2017). While porpoise strandings are reasonably common along the mainland Atlantic coast of the Iberian Peninsula, records of stranded porpoises are extremely rare in the Macaronesian archipelagos.

Spain

Lens (1997) included a table of documented porpoise strandings in Spain during 1978–1994. The great majority (86 out of 100) occurred along the western Galician coast, with a further 12 along the Biscay coasts of Galicia, Asturias, Cantabria and the Basque country. Note that four out of five records of porpoises stranded in the vicinity of A Coruña in Galicia in April 1981 (Rey & Cendrero 1982) were missed out. From 1990 to 2013, porpoises were the third most frequently recorded cetacean species in strandings in Galicia, with a total of 193 individuals recorded. There was a clear seasonal pattern, with strandings being most numerous in late winter to early spring, but no apparent long-term trend (López et al. 2002, Saavedra et al. 2017).

During 2008–2020, 55 stranded porpoises were recorded along the Atlantic coast of Andalucía. Of these, 13 were stranded during 2019–2020 (Anon. 2019, 2020, 2021). Rojo-Nieto et al. (2011) mentioned two harbour porpoises stranded in Cádiz (in 2004 and 2008) among a total of 1198 marine mammal strandings along the southern coast of Spain and the North African coast (between the Spanish enclaves of Ceuta and Melilla) during 1991–2008. Cabezón et al. (2004) reported on the parasites of a porpoise stranded on the Mediterranean coast of Andalucía. Bellido et al. (2006) reported a porpoise floating (alive) off the coast and later stranded (dead) at Benalmádena (Málaga), noting that this was the first porpoise reported stranded in Alboran Sea since the individual stranded in 1981 and reported by Rey and Cendrero (1982). Three stranded porpoises were reported from the Mediterranean coast between Gibraltar and the Alboran Sea between 2008 and 2022 (Anon. 2022a)[3], although the only individual stranding records from this area during that period appear to be from 2009 and 2010 (Anon. 2009a, 2010). There were no records of strandings of harbour porpoises along the French Mediterranean coast during 1970–1994 (Collet 1995). Kaddouri et al. (2023) reported three porpoise strandings on the northwest coast of Morocco during 2016–2021, including one from the Strait of Gibraltar. It is unclear whether these animals would have been from the Iberian population or the African population. The only known stranding of a porpoise in the Canary Islands occurred in 2006 on the Island of Fuerteventura (V. Martín, Pers. Comm., Díaz-Delgado et al. 2018, Puig-Lozano et al. 2020).

Portugal

As previously mentioned, Sequeira and Ferreira (1994) reported a decline in porpoise strandings between 1977 (when strandings monitoring began in Portugal) and the late 1980s. According to Sequeira (1996; see also Sequeira et al. 1996, 1997), 86% (69) of the 80 porpoise strandings in

[3] The same source gives the total for the Atlantic and Mediterranean coasts together as 57, so the numbers do not add up.

Portugal recorded between the nineteenth century and 1994 occurred along the northern and central zones of the west coast, especially around Aveiro and Figueira da Foz (where 67% of strandings occurred). Strandings were thus relatively infrequent south of Lisbon and along the south coast. There was a clear seasonal pattern with more strandings during January to April. Between 2000 (when a new strandings network was established in central and northern Portugal) and 2016, 4021 stranded cetaceans were reported from the Portuguese coast. On average, 17 porpoises were stranded on Portuguese mainland coasts each year, with a generally increasing trend between 2002 and 2014 (43 stranded individuals were recorded in the latter year) (Vingada & Eira 2018). We have no information on search effort, but a gradual increase in effort since the establishment of the network is plausible and might explain the trend in numbers reported. Strandings data on porpoises were also compiled by Gonçalves de Sousa (2010) for the entire mainland coast of Portugal during 1979–2009, who noted that there were 251 porpoise strandings recorded along the Portuguese mainland coast, making up 7.13% of cetacean strandings during that period. Ângelo (2020) compiled strandings data for the west coast of Portugal during 2010–2019 and referred to 278 porpoise strandings during this period. Torres-Pereira et al. (2023a) summarised porpoise strandings in Portugal during 2000–2020, noting an increase in numbers stranded over the years, especially in 2015–2020, and that most strandings occurred in April to June.

In southern Portugal, the regional network established in 2010 recorded on average three to four porpoises strandings per year up to 2019 and since then a drastic decrease in numbers (a single-stranded porpoise was reported in the Algarve from 2020 to 2022) (A. Marçalo, Pers. Obs.). Compiling this information, there are some minor discrepancies in annual totals (so we used the highest values) but the time series clearly illustrates a marked increase in the number of strandings during the 2010s (Figure 10).

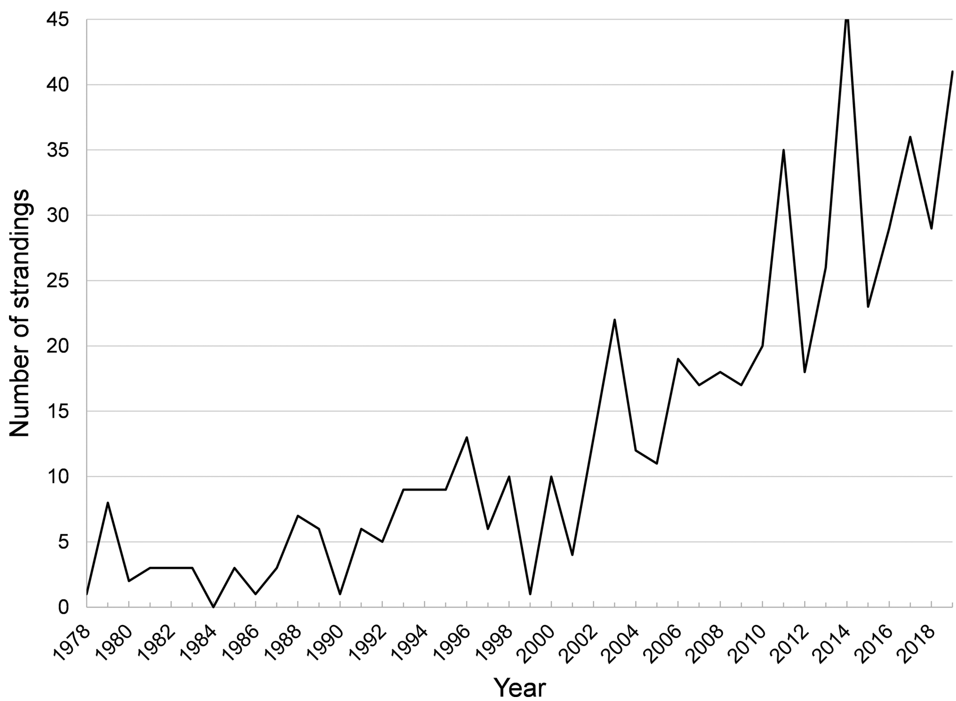

Figure 10 Number of porpoise strandings per year registered along the mainland coast of Portugal from 1978 to 2019. Compiled from Gonçalves de Sousa (2010), Vingada and Eira (2018), Ângelo (2020) and unpublished data from the southern strandings network (A. Marçalo pers. comm.).

In 2004, the stranding of an adult female was recorded on the island of Terceira in the Azores (Barreiros et al. 2006). The origin of this animal remains unknown.

Life history

The most up-to-date life history parameter data for porpoises from the Iberian Peninsula concern the area from the northern limit of Galicia (Northwest Spain) (43°3′N, 7°2′W) to north-central Portugal as far south as Nazaré (39°5′N, 9°2′W) and derive mainly from individuals stranded between 1990 and 2010. A few bycaught animals brought to land during this period were also included (Read et al. 2013, 2020, Read 2016). We refer to this region throughout this section as the Northwest Iberian Peninsula (NWIP). Some historical information for Portugal is available in Sequeira (1996) for the period 1981–1994. In Tables 3A and 3B, published information on the life history parameters of both sexes is summarised for various populations of harbour porpoises in the North Atlantic. Similar (but less extensive) tables may be found in Lockyer and Kinze (2003) and Read (2016). Information from the latter tables was included only when it could be checked against the original source.

A degree of caution is needed when interpreting these life history results. The maximum age of animals in a population will be inversely related to the mortality rate but the likelihood of detecting the oldest animals will be lower if sample size is smaller. Most recent studies on the life history of small cetaceans are based on stranded animals. Given that strandings monitoring networks usually work with limited resources, the oldest age that is determined may also depend on the sample selection strategy. Determining sexual maturity in females should be straightforward although it is important to examine both ovaries for the presence of a *corpus albicans* or *corpus luteum* (even if in harbour porpoise, ovulation and embryonic implantation are normally associated with the left ovary and left uterine horn, respectively) (Harrison 1971, Karakosta et al. 1999). In males, the assessment of maturity is normally based on the histological examination of the testes although Lockyer (1995b) proposed the use of a combined testis weight >200 g as an indicator of maturity.

Age estimation is usually based on counting growth layer groups in teeth, and it is usual for tooth sections to be read by at least two persons (Read et al. 2018) but some studies have used body length as a proxy for age and their conclusions about age at maturity should be treated with caution. To reliably estimate the age and length at sexual maturity (normally taken to be the age and length at which 50% of individuals are mature), a good sample size is needed for animals of ages and sizes closest to the values at which they reach maturity. Evidently, this may be difficult to achieve, because age and size at maturity are initially unknown and, in any case, sample size may be limited by the availability of stranded animals and/or the resources needed to process them. Hence, the overall sample size is not a good indicator of the accuracy and precision of the estimates that can be obtained. In addition, the resulting estimates may depend on what model (if any) is fitted (e.g. logistic regression has an inbuilt symmetry assumption which can drive the resulting estimate) and whether it is fitted to individual data or data that have been grouped by age or length class. Confidence limits and sample sizes are not always reported. Some authors (e.g. Ólafsdóttir et al. 2003) have reported several alternative metrics for Length and Age at Sexual Maturity (LSM and ASM, respectively), including use of the mean age (plus one) of pubertal animals (both sexes), the mean length and age of first-time ovulators, the estimated age at which the number of ovarian scars (corpora albicantia plus corpus luteum) first exceeds zero (based on a linear regression of the total number of scars versus age), and the DeMaster method (DeMaster 1978), which uses the mean proportion of mature animals across all age classes, corrected for the decline in abundance of older age classes (see also DeMaster 1984 for a review of methods).

Assessment of age at physical maturity in cetaceans should ideally be based on the fusion of epiphyseal plates in the vertebrae (e.g. Ridgway & Harrison 1999), but in practice it is often based on fitting a growth curve to determine the asymptotic body length (e.g. Murphy et al. 2009). While it is clear from some studies (e.g. Mead et al. 1982) that growth in cetaceans continues beyond physical

Table 3A Life History Parameters of Female Harbour Porpoises in the Atlantic Ocean

Area and Time Period	Source	Maximum Length (cm)	Maximum Age (yrs)	Sexual Maturity Length (cm)	Sexual Maturity Age (yrs)	Asymptotic Length (cm)	Age at Asymptotic Length (yrs)	Pregnancy Rate (Presence of Foetus)
NWIP (1990–2010)	1	202 (*N* = 125)	18 (*N*=71)	169 (*N*=60)	5.5 (*N*=60)	185 (*N*=60)	10 (*N*=60)	0.54 (*N* = 13)
Atlantic Spain (1978–1994)	2	202 (*N*=32)	n/a	n/a	n/a	n/a	n/a	n/a
Galicia, NW Spain (1990–1999)	3	202 (*N*=35)	9	166 (*N*=35)	3	n/a	n/a	n/a
Portugal (1981–1994)	4	208 (*N* = 22)	n/a	n/a	n/a	n/a	n/a	n/a
Portugal (2002–2016)	25	219 (77)	–	168.9 (*N*=77)	–	–	–	0.68[b]
Portugal (2017–2019)	26	199 (30)	–	–	–	–	–	
Scotland (1992–2005)	5	173 (*N*=289)	20 (*N*=170)	138.8 (CI=135.9–141.6) (*N*=190)	4.35 (CI=3.93–4.71) (*N*=144)	GAM: 158.4 (SE=2.69, CI=153.1–163.8) Gompertz: 160.7 (CI=157.7–163.8 cm) (*N*=170)	12 (*N*=170)	i. 0.34 (CI=0.17–0.52) (*N*=29) ii. 0.40 (CI=0.26–0.55) (*N*=42)
British Isles (1985–1994)	6	189 (*N*=128)	22 (*N*=96)	n/a	n/a	160 (*N*=96)	8 (*N*=96)	n/a
England (North Sea MU) (1990–1999)	7	172 (*N*=79)	22 (*N*=72)	138.90 (SE=1.46) (*N*=75)	3.8 (SE=0.23) (*N*=68)	155.37 (SE=1.95) (*N*=72)	7.21 (*N*=72)	0.26 (*N*=19)
England (North Sea MU) (2000–2013) [a](2000–2012)	7	180 (*N*=109)	15 (*N*=51)	139.18 (SE=1.44) (*N*=90)	4.8 (SE=0.31) (*N*=49)[a]	155.37 (SE=1.95) (*N*=51)[a]	11.66 (*N*=51)[a]	0.30 (*N*=23)
England and Wales (Celtic and Irish Seas MU (1990–1999)	7	191 (*N*=127)	15 (*N*=107)	146.56 (SE=1.71) (*N*=121)	3.8 (SE=0.23) (*N*=102)	162.94 (SE=1.95) (*N*=104)	7.21 (*N*=104)	0.68 (*N*=25)
England and Wales (Celtic and Irish Seas MU) (2000–2013) [a](2000–2012)	7	189 (*N* = 269)	21 (*N*=88)	146.94 (SE=1.32) (*N*=199)	4.8 (SE=0.31) (*N*=86)[a]	162.94 (SE=1.95) (*N*=87)[a]	11.66 (*N*=87)[a]	0.54 (*N*=35)
Ireland (2001–2003)	8	175 (*N*=27)	11 (*N*=21)	S coast: >150 Irish Sea: >140	S coast: >5 Irish Sea: 3.5–5	n/a	n/a	0.4 (*N*=5)

Table 3A (*Continued*) Life History Parameters of Female Harbour Porpoises in the Atlantic Ocean

Area and Time Period	Source	Maximum Length (cm)	Maximum Age (yrs)	Sexual Maturity Length (cm)	Sexual Maturity Age (yrs)	Asymptotic Length (cm)	Age at Asymptotic Length (yrs)	Pregnancy Rate (Presence of Foetus)
Denmark (1838–1998) [a](1996–1998)	9	189	23	143[a]	3.5 (*N*=25)[a]	160	n/a	n/a
Germany (Schleswig Holstein - North Sea) (1990–2016)	10	n/a	22 (*N*=311)	n/a	4.95 (95% CI= 4.2–4.8) (*N*=311)	n/a	n/a	n/a
Germany (Schleswig Holstein - western Baltic Sea) (1990–2016)	10	n/a	22 (*N*=215)	n/a	4.95 (CI= 4.2–4.8) (*N*=215)	n/a	n/a	n/a
The Netherlands (1955–1976)	11	186 (*N*=62)	12 (*N*=34)	150 (*N*=19)	6 (*N*=18)	~150 (*N*=34)	~6 (*N*=34)	n/a
The Netherlands (2001–2003)	8	160 (*N*=19)	12 (*N*=14)	>130 (*N*=19)	5 (*N*=14)	n/a	n/a	0.11 (*N*=9)
The Netherlands (2006–2019)	12	175 (*N*=337)	24 (*N*=154)	n/a	4 (CI=3.47–4.48) (*N*=154)	n/a	n/a	0.34 (CI= 0.26–0.43) (*N*=119)
France (Bay of Biscay) (1970–1994)	13	186	n/a	n/a	n/a	n/a	n/a	n/a
France (French English Channel) (1970–1994)	13	190	n/a	n/a	n/a	n/a	n/a	n/a
France (S North Sea, W and E Channel, and Bay of Biscay combined) (2001–2003)	8	192 (*N*=14)	24 (*N*=9)	n/a	n/a	n/a	n/a	n/a
France (North Sea MU) (1990–2015)	14	168	20 (*N*=130)	n/a	3.4 (*N*=130)	n/a	n/a	n/a
France (Celtic Sea MU) (1990–2015)	14	196	24 (*N*=81)	n/a	2.5 (*N*=81)	n/a	n/a	n/a

(*Continued*)

Table 3A (*Continued*) Life History Parameters of Female Harbour Porpoises in the Atlantic Ocean

Area and Time Period	Source	Maximum Length (cm)	Maximum Age (yrs)	Sexual Maturity Length (cm)	Sexual Maturity Age (yrs)	Asymptotic Length (cm)	Age at Asymptotic Length (yrs)	Pregnancy Rate (Presence of Foetus)
West Greenland (1988–89, 1995)	15	166 (*N*=71)	14 (*N*=84)	142 (*N*=71)	3.6	154±2.6	n/a	n/a
West Greenland (2009)	16	n/a	n/a	n/a	3.5 (SE=0.03) (*N*=60)	n/a	n/a	n/a
Norway (1988–1989)	17	168 (*N*=35)	>8 (*N*=35)	n/a	4 (*N*=35)	155.9 (*N*=35)	n/a	n/a
Norway (mainly northern Norway) (2016–2017)	18	173 (*N*=58)	7 (*N*=48)	n/a	4.3 (CI±0.6) (*N*=48)	165.2 (CI= 155.7–176.9) (*N*=58)	n/a	0.85 (CI±0.16) (*N*=20)
Iceland (1991–1997)	19	174 (*N*=474)	20 (*N*=354)	1. 138 (*N*=72) 2. 146 cm 3. 147.6 cm (*N*=30)	1. 2.5 (*N*=62) 2. 3.2 (*N*=269) 3. 2.81 (*N*=21) 4. 2.1 (*N*=51) 5. 4.4 (*N*=293)	160.1 (*N*=314)	n/a	0.98 (*N*=74)
Gulf of Maine/Bay of Fundy (1989–1993) [a](1985–1993)	20, 21[a]	168 (*N*=203)[a]	17 (*N*=109)	n/a	3.36 (SD=0.13) (*N*=99)	158 (CI±1.56) (*N*=203)[a]	~7 (*N*=203)[a]	0.93 (*N*=14)
Bay of Fundy (1969–1973)	22	>165 (*N*=44)	9 (*N*=44)	147 (CI±1.7) (*N*=46)	3.97 (CI± 0.49) (*N*=37)	163 (CI±8.4) (*N*=44)	n/a	0.89 (CI± 0.14) (*N*=19)
Bay of Fundy (1985–1988)	22	~163 (*N*=116)	10 (*N*=116)	143 (CI±0.83) (*N*=108)	3.44 (CI±0.36) (*N*=108)	155 (CI±3.5) (*N*=116)	~6–7 (*N*=116)	0.86 (CI±0.09) (*N*=50)
Newfoundland (1990–1991)	23, [a]24	162 (*N*=35)[a]	9 (*N*=35)	146.4 (SE=0.03) (*N*=32)[a]	3.1 (SE=0.07) (*N*=32)[a]	156.3 (SE=2.9) (*N*=33)	>4 (*N*=33)	0.76 (SE=0.1) (*N*=17)[a]

Table 3B Life History Parameters of Male Harbour Porpoises in the Atlantic Ocean

Area and Time Period	Source	Maximum Length (cm)	Maximum Age (yrs)	Sexual Maturity Length (cm)	Sexual Maturity Age (yrs)	Asymptotic Length (cm)	Age at Asymptotic Length (yrs)
NWIP (1990–2010)	1	189 (*N*=135)	19 (*N*=77)	151 (*N*=47)	3.8 (*N*=47)	162 (*N*=47)	10 (*N*=47)
Atlantic Spain (1978–1994)	2	176 (*N*=27)	n/a	n/a	n/a	n/a	n/a
Galicia, NW Spain (1990–1999)	3	180 (*N*=42)	9	155 (*N*=9)	5	n/a	n/a
Portugal (1981–1994)	4	175 (*N*=15)	n/a	n/a	n/a	n/a	n/a
Scotland (1992–2005)	5	170 (*N*=320)	20 (*N*=176)	132.2 (CI=129.1–135.6) (*N*=145)	5 (CI=4.03–5.88) (*N*=105)	GAM: 147.2 (SE=1.66, CI=143.95–150.5) Gompertz: 148.3 (CI=146.3–150.0 (*N*=180)	12 (*N*=180)
British Isles (1985–1994)	6	163 (*N*=144)	24 (*N*=114)	>130 (*N*=114)	>3 (*N*=114)	145 (*N*=114)	8 (*N*=114)
England (North Sea MU) (1990–1999)	7	153 (*N*=89)	18 (*N*=83)	133.27 (SE=1.33) (*N*=64)	3.56 (SE=0.25) (*N*=62)	140.94 (SE=1.64) (*N*=83)	5.72 (*N*=83)
England (North Sea MU) (2000–2013) [a](2000–2012)	7	161 (*N*=146)	16 (*N*=50)	129.47 (SE=1.29) (*N*=97)	3.62 (SE=0.26) (*N*=45)[a]	140.94 (SE=1.64) (*N*=49)[a]	7.62 (*N*=49)[a]
England and Wales (Celtic and Irish Seas Management Unit) (1990–1999)	7	171 (*N*=129)	18 (*N*=110)	138.73 (SE=1.50) (*N*=92)	3.56 (SE=0.25) (*N*=78)	146.50 (SE=1.60) (*N*=109)	5.72 (*N*=109)
England and Wales (Celtic and Irish Seas Management Unit) (2000–2013) [a](2000–2012)	7	181 (*N*=271)	15 (*N*=84)	133.46 (SE=1.24) (*N*=164)	3.62 (SE=0.26) (*N*=66)[a]	146.50 (SE=1.60) (*N*=83)[a]	7.62 (*N*=83)[a]
Ireland (2001–2003)	8	157 (*N*=17)	7.5 (*N*=14)	S and W coasts: 134–144 Irish Sea: 131–146	S and W coasts: 3–7 Irish Sea: 4–8	n/a	n/a
Denmark (1838–1998) [a](1996–1998)	9	167	23	135 (*N*=135)[a]	3–4 (*N*=135)[a]	145	n/a
Germany (Schleswig Holstein - North Sea) (1990–2016)	10	n/a	n/a	n/a	n/a	n/a	n/a

(*Continued*)

Table 3B (*Continued*) Life History Parameters of Male Harbour Porpoises in the Atlantic Ocean

Area and Time Period	Source	Maximum Length (cm)	Maximum Age (yrs)	Sexual Maturity Length (cm)	Sexual Maturity Age (yrs)	Asymptotic Length (cm)	Age at Asymptotic Length (yrs)
Germany (Schleswig Holstein - western Baltic Sea) (1990–2016)	10	n/a	n/a	n/a	n/a	n/a	n/a
The Netherlands (1955–1976)	11	151 (*N*=43)	12 (*N*=20)	135 (*N*=46)	5 (*N*=20)	~130 (*N*=20)	~4 (*N*=20)
The Netherlands (2001–2003)	8	n/a	n/a	n/a	n/a	n/a	n/a
The Netherlands (2006–2019)	12	n/a	n/a	n/a	n/a	n/a	n/a
France (Bay of Biscay) (1970–1994)	13	168	n/a	n/a	n/a	n/a	n/a
France (French English Channel) (1970–1994)	13	183	n/a	n/a	n/a	n/a	n/a
France (S North Sea, W and E Channel, and Bay of Biscay combined) (2001–2003)	8	165 (*N*=17)	14 (*N*=12)	n/a	n/a	n/a	n/a
France (North Sea MU) (1990–2015)	14	170	14 (*N*=164)	n/a	n/a	n/a	n/a
France (Celtic Sea MU) (1990–2015)	14	183	11.5 (*N*=89)	n/a	n/a	n/a	n/a
West Greenland (1988–89, 1995)	15	158 (*N*=81)	17 (*N*=94)	127 (123–130)	2.45	143±1.7	n/a
West Greenland (2009)	16	n/a	n/a	n/a	3.1 (SE=0.08) (*N*=29)	n/a	n/a
Norway (1988–1989)	17	147 (*N*=41)	>8 (*N*=41)	n/a	3 (*N*=41)	142.3 (*N*=41)	n/a
Norway (mainly northern Norway) (2016–2017)	18	158 (*N*=75)	12 (*N*=75)	n/a	2–3 (*N*=75)	149.0 (CI=145.4–152.8) (*N*=75)	n/a
Iceland (1991–1997)	19	165 (*N*=794)	16 (*N*=615)	1. 135.6 (*N*=33) 2. 135	1.2.9 (*N*=21) 2. 1.9 (*N*=493) 3. 2.6 (*N*=526)	149.6 (*N*=497)	n/a

Table 3B (*Continued*) Life History Parameters of Male Harbour Porpoises in the Atlantic Ocean

Area and Time Period	Source	Maximum Length (cm)	Maximum Age (yrs)	Sexual Maturity Length (cm)	Sexual Maturity Age (yrs)	Asymptotic Length (cm)	Age at Asymptotic Length (yrs)
Gulf of Maine/Bay of Fundy (1989–1993) [a](1985–1993)	20, 21[a]	157 (*N*=198)[a]	15 (*N*=123)	n/a	>3 (3–4) (*N*=31)	143 (CI±1.25) (*N*=198)[a]	~5 (*N*=198)[a]
Bay of Fundy (1969–1973)	22	~150 (*N*=56)	10 (*N*=56)	n/a	n/a	146 (CI±4.1) (*N*=56)	~7–8 (*N*=56)
Bay of Fundy (1985–1988)	22	~153 (*N*=121)	10 (*N*=121)	n/a	n/a	144 (CI±3.5) (*N*=121)	~6–7 (*N*=121)
Newfoundland (1990–1991)	23, 24[a]	155.5 (*N*=59)[a]	12 (*N*=59)	135.1 (SE=0.02) (*N*=59)[a]	3 (*N*=59)[a]	142.9 (SE=1.2) (*N*=59)	>4 (*N*=59)

Sources: 1: Read (2016), 2: Lens (1997), 3: López Fernández (2003), 4: Sequeira (1996), 5: Learmonth et al. (2014), 6: Lockyer (1995a), 7: Murphy et al. (2020), 8: Learmonth et al. (2004), 9: Lockyer and Kinze (2003), 10: Kesselring et al. (2017), 11: van Utrecht (1978), 12: IJsseldijk et al. (2021), 13: Collet (1995), 14: Rouby (2018), 15: Lockyer et al. (2001), 16: Heide-Jørgensen et al. (2011), 17: Bjørge et al. (1991), 18: Unpublished data cited in NAMMCO/IMR 2019, 19: Ólafsdóttir et al. (2003), 20: Read and Hohn (1995), 21: Read and Tolley (1997), 22: Read and Gaskin (1990), 23: Richardson et al. (2003), 24: Richardson (1992), 25: Camarão (2017), 26: Oliveira (2020).

Notes: Camarão (2017) weighed and measured ovaries of 77 porpoises but it is not stated how many of these animals were mature and the pregnancy rate may have been based on a sub-sample of the mature animals. Learmonth et al. (2014) used two different models to estimate asymptotic length, a Generalised Additive Model (GAM) and a Gompertz model. Lockyer and Kinze (2003) assembled a range of different data sources, the earliest dating back to 1838, but it is not always clear which animals contributed to life history parameter estimates. Rouby (2018) used a regression model with Weibull distribution to estimate ASM. The two pregnancy rates cited were based on excluding mature females stranded during (i) April to September and (ii) 26 May to 12 September. Ólafsdóttir et al. (2003) reported several estimates of LSM and/or ASM, some applicable to both sexes: (1) based on the mean age plus one of pubertal animals, (2) the length or age at which 50% of animals are mature, (3) mean length at first ovulation, (4) linear regression method to estimate the age at which the number of ovarian scars first exceeds zero, (5) the DeMaster method (DeMaster 1978), based on the mean of the proportion of mature animals in each age class, corrected for mortality. For some estimates, standard deviation (SD), standard error (SE) and/or 95% confidence intervals (CI) were available and are reported here. Sample sizes (*N*) are reported when available.

NWIP, northwest Iberian Peninsula; S, south or southern; W, west; E, East; MU, management unit; n/a, not available.

[a] Results deriving from a different time period or separate study as detailed in columns 1 or 2.

[b] This is the proportion of mature females with a corpus luteum in the ovaries. The sample size is unknown.

maturity, there appears to be no consensus in the literature about the relationship between asymptotic size and size at physical maturity, with the latter being variously equated to 90%, 95% or (by default) 100% of the former (Aguilar & Lockyer 1987, Pribanić et al. 2000, Murphy et al. 2020, Betty et al. 2022). In addition, fitting different types of growth curve may result in different estimates of asymptotic length (e.g. Richardson et al. 2003, Betty et al. 2022). Most authors used a Gompertz model (although there are different variants of this) and a few used a von Bertalanffy model (e.g. Ólafsdóttir et al. 2003). Where age at physical maturity is reported, it is usually age at asymptotic length; where it was not reported but the growth curve was provided, we have estimated it from the growth curve.

Pregnancy rate may be underestimated if animals that died during the conception period are included in the calculations, because the foetus, if present in such animals, will be very small and hence difficult to detect. A further issue with using data from stranded animals is that pregnancy rate is likely to be underestimated due to the high proportion of sampled females that were in a poor state of health. Where sample sizes permit, this bias can be avoided by basing estimates of pregnancy rate on animals that died due to physical trauma (e.g. fishery bycatch, collisions with boats or fatal interactions with bottlenose dolphins), on the assumption that their health state is more likely to have been representative of the living population. An additional issue with annual pregnancy rate values is that some authors divide the proportion of mature females that are pregnant by the length of the gestation period, typically 10–12 months in porpoises, although where the seasonality of breeding is consistent (as found by Learmonth et al. 2014 in Scotland), implying that the breeding cycle effectively lasts 1 year, such an adjustment is not needed (Read 2016). One possible means to ensure that observed differences are not artefacts of differing methodologies would be to undertake new combined analyses of raw data from different Assessment Units (see ICES 2014a), if available, as done by Murphy et al. (2020) to compare North Sea and Celtic Sea porpoises.

Size, age, growth and maturation

Maximum reported length in animals for the Iberian Peninsula is greater than that in the other populations, as are the lengths at sexual and physical maturity (see Table 3). Vaz (2015) noted that in addition to a larger body length, the cranium of porpoises from Portuguese waters was longer and wider than that of porpoises from the north of the Bay of Biscay. The large body size of Iberian porpoises is well known (e.g. Smeenk et al. 1992, Donovan & Bjørge 1995, Sequeira 1996, López Fernández 2003, Murphy et al. 2020). Maximum lengths recorded were 219 cm for females and 189 cm for males (Sequeira 1996, Read et al. 2013, Camarão 2017). Two of the three porpoises reported stranded on the northwest coast of Morocco during 2016–2021 were measured, a female of 150 cm and a male of 210 cm (Kaddouri et al. 2023). It is not clear whether these would have been African or Iberian animals and the length reported for the male is unusually large. At the opposite extreme, the maximum length for females in the Netherlands reported by Learmonth et al. (2004) was 160 cm and the maximum length reported for males in Norway by Bjørge et al. (1991) was 147 cm.

Various factors operating over different time-scales are expected to influence body size and other life history characteristics in harbour porpoises. The reproductive strategy of the males involves allocating a high proportion of available energy resources to sexual maturation and reproduction, maturing at a relatively small body size and maintaining a large testicular mass during the breeding period, while females need to reach a certain size to be able to carry their offspring. Males retain more paedomorphic features than females (Murphy et al. 2020). While the explanation for the larger body size of Iberian porpoise remains unclear, it may be relevant that their upwelling habitat is characterised by both high productivity and high variability in productivity. Murphy et al. (2020) proposed that the larger body size of Iberian porpoise could be an adaptation to periods of shortage of resources (after Perrin 1989, Ferguson & Larivière 2008, Ferguson et al. 2018). In their review of interspecific variation in body size in terrestrial mammalian carnivores, Ferguson and Larivière (2008) observed that:

species living in highly seasonal environments were associated with larger home ranges and low density that in turn selected for larger body mass and greater sexual size dimorphism. Thus, spacing behaviour provides an important evolutionary link explaining interspecific body size variation.

The maximum age recorded for porpoises in the NWIP was 18 years for females and 19 years for males (Read 2016). One animal of undetermined sex reached 21 years of age. These ages fall within the (admittedly wide) range of maximum ages reported for other areas of the North Atlantic. Maximum ages for females in the Northeast Atlantic ranged from 7 in Norway (North Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research 2019) to 24 in France (Learmonth et al. 2004), while those for males ranged from 7.5 in Ireland (Learmonth et al. 2004) to 24 in the British Isles (Lockyer 1995a). As noted above, estimated maximum age may depend on the sample size and sample selection strategy, and several of the above-mentioned studies had small sample sizes (Tables 3A and B).

As stated above, in harbour porpoise, ovulation normally occurs only in the left ovary, which tends to be larger and heavier and contain more ova (e.g. Karaskota et al. 1999, Learmonth et al. 2014). This difference has been confirmed in Iberian porpoises (Camarão 2017, Oliveira 2020). Camarão (2017) also reported that corpora albicantia were found four times as frequently in the left ovaries of mature females as in the right ovaries (and never only in the right ovary).

Female Iberian porpoises reached sexual maturity at a length of 169 cm (range of lengths for mature females: 161–202 cm; range for pregnant females: 176–202 cm) (Read et al. 2013, Read 2016). Camarão 2017 reported that mature females ranged in length from 159 cm to 219 cm, with 50% of females reaching sexual maturity at a length of 168.9 cm. The length at maturity was the highest reported for the North Atlantic, the lowest values being those reported for Greenland and various sites along the Atlantic coast of North America (all less than 170 cm). This excludes the 130 cm maximum length reported for the Netherlands by Learmonth et al. (2004), which was based on a very small sample size. Both previous and subsequent studies in the Netherlands based on larger sample sizes gave maximum length values more in line with those for nearby countries such as France, the UK and Denmark (Table 3A).

ASM for female Iberian porpoises was estimated to be around 5.5 years old (Read 2016), among the highest values reported for the North Atlantic and similar to results for the Netherlands: 5 years old according to Addink et al. (1995) and 6 years old according to van Utrecht (1978). The lowest reported estimate of ASM was 3.1 years old in Newfoundland (Richardson et al. 2003). It is not clear why maturation should be relatively late in females from the NWIP; there is no obvious selective advantage. Ideally, the veracity of this finding requires confirmation using larger sample sizes. It should be noted that, based on a much smaller sample size, López Fernández (2003) estimated the age at maturity for females in Galicia to be 3 years old. The multiple estimates of ASM (and of LSM) presented by Ólafsdóttir et al. (2003) for porpoises of both sexes in Iceland illustrate the sensitivity of estimates to the methodology used. The lack of confidence limits for many estimates is also an issue when evaluating differences between porpoises from different areas.

Length at sexual maturity for male Iberian porpoises was estimated to be 151 cm (Read 2016), with mature males ranging in length from 154 cm to 171 cm (Read et al. 2013). This is the highest value reported for the North Atlantic, with the lowest being for Greenland (127 cm, Lockyer et al. 2001). Males mature at 3.8 years old (Read 2016). This falls within the range reported elsewhere in the North Atlantic, where values range from 2.45 in west Greenland (Lockyer et al. 2001) to 5.7 in Scotland (Learmonth et al. 2014).

Growth models indicated that the asymptotic length of Iberian males was 162 cm and that of females was 185 cm (Read 2016), higher than that reported elsewhere in the North Atlantic, with the lowest values being those for the Netherlands (150 cm for females and 130 cm for males; van Utrecht 1978). The age at asymptotic length for Iberian porpoises (approximately 10 years old) was also among the highest (Read 2016; Table 3). Values for both sexes ranged from around 4 years old in the

Netherlands and Newfoundland (van Utrecht 1978, Richardson et al. 2003) to around 12 years old in Scotland (Learmonth et al. 2014); none of the studies reviewed reported true physical maturity.

Over 85% of stranded (and a few bycaught) animals in the NWIP for which age was determined were ≤10 years old, and over 60% were ≤3 years old (Read et al. 2013). Similar age distribution patterns have been observed in the Northeast Atlantic. Based on a large sample size of 645 necropsied porpoises stranded and bycaught along the English and Welsh coasts between 1990 and 2012, Murphy et al. (2020) found that around 80% were ≤5 years old and only 5% were aged 12 years or older. The relative rarity of old animals was also apparent in German waters of the North Sea and Baltic Sea, where 90% of porpoises did not live longer than 12 and 9.6 years, respectively (Kesselring et al. 2017); Danish waters, where less than 5% of porpoises lived to ages beyond 12 years (Lockyer & Kinze 2003); Dutch waters, where 75% of mature females did not live longer than 8.5 years (IJsseldijk et al. 2021a); and Icelandic waters, where up to 90% of the porpoises examined were less than 6 years old (Ólafsdóttir et al. 2003).

Life history traits can vary both over time and geographically, due to both intrinsic and extrinsic factors (e.g. Murphy et al. 2020). Thus, by obtaining more recent life history data from stranding and bycatch since the 2010s, evidence of changes in life history parameters due to pressures such as anthropogenic mortality (e.g. bycatch) may be detectable. Work is still needed to analyse more recent samples from the NWIP.

Murphy et al. (2020) compared length-related life history traits of porpoises from the Celtic and Irish Seas Management Unit (CIS MU) during 1990–1999 and 2000–2013. Porpoises from the CIS MU have a larger body size than animals from other areas in the Northeast Atlantic, which may be inherited from large-sized Iberian porpoises. The admixture zone between the two ecotypes is restricted to the northern part of the Bay of Biscay and southern parts of the Celtic Sea and English Channel (Fontaine et al. 2017). At any given age, porpoises were larger in the 1990s compared to the 2000s and 2010s, and there was also a decline in the Gompertz growth rate parameter, both trends being more evident in females. Declines were also seen in the length at sexual maturity of males and in the occurrence of the “larger-sized morphotype”. The authors suggested that these changes may reflect a decrease in movements of Iberian porpoises towards the CIS MU, although acknowledging that other factors could also be at play. If correct, this could be related to the apparent decline in the abundance of Iberian porpoises between 1990 and 2015, as proposed by Ben Chehida et al. (2023) to explain the observed reduction of genetic diversity. Such a decline could have resulted in fewer animals migrating towards northern waters and hence a reduced contribution to the genetic pool of animals in the CIS MU.

Causes of death, sex ratio, age structure and mortality rate

The quality of information available on causes of death depends on several factors, including whether strandings networks have funding for necropsies, access to veterinary expertise and the frequency with which a species is stranded. Ideally, cause of death should be interpreted in the context of a complete gross, histological and histopathological examination of the carcass and sampled tissues. Publications providing guidelines for necropsy protocols and diagnosis of cause of death in cetaceans include Kuiken (1996), IJsseldijk et al. (2019) and Jauniaux et al. (2019).

Among the most important reported causes of death in porpoises in the Northeast Atlantic are bacterial and parasitic pneumonia and fishery bycatch. Typically, bycatch mortality in porpoises can be detected based on the presence of superficial incisions (net cuts), encircling imprints (mainly on the head but also on the flippers and body) and changes in the lungs (pulmonary oedema, emphysema and atelectasis), while circumstantial evidence includes recently ingested prey, reddish or bulging eyes, congestion and disseminated gas bubbles. Post-mortem injuries including stab wounds, amputations and abdominal cuts may have been the result of freeing the animal from a net or trying to sink the carcass. It is also important to be able to exclude other possible causes of death (as determined from other significant pathological findings) (Puig-Lozano et al. 2020, IJsseldijk et al. 2021b). IJsseldijk et al. (2022) reported that infectious diseases (32%) were the largest cause of death category in porpoises stranded in the Netherlands, while bycatch accounted for a further 17% of mortalities. In Sweden, fishery bycatch was the most important cause of death (31%) with diseases (mostly pneumonia) accounting for a further 21% (Neimanis et al. 2022). Cases of starvation in adults and in neonates separated from their mothers, as well as porpoise deaths caused by non-infectious diseases such as congenital anomalies, perinatal complications (dystocia and stillbirths), gastritis and neoplasms, have been described in the UK (Kirkwood et al. 1997), the Netherlands, Belgium, Germany (van Elk et al. 2019) and Sweden (Neimanis et al. 2022). Deaths due to bottlenose dolphin attacks are common in coastal waters of northeast Scotland, where there is a resident bottlenose dolphin population (Ross & Wilson 1996, Davison & ten Doeschate 2020). Acoustic monitoring of porpoise distribution in this area suggested that they tended to avoid spatio-temporal overlap with bottlenose dolphins (Williamson et al. 2022).

In the NWIP, bycatch is a commonly diagnosed cause of death in harbour porpoise (Sequeira & Ferreira 1994, Read et al. 2013, Read 2016). In a compilation of findings from necropsies of cetaceans stranded in Galicia during 2002–2003, results from three porpoises were described. One was diagnosed as bycaught, and no additional findings were mentioned. The second presented parasitic pneumonia, and the third animal had severe pulmonary congestion and pulmonary oedema with lungworms present, also several gastric ulcers and lesions consistent with bycatch (Alonso et al. 2004).

There is little published information on the sex ratio in the Iberian porpoise population. Read et al. (2013) found the sex ratio in stranded animals from the NWIP to be close to parity at 1:1.07 females to males (not significantly different from 1:1). Lens (1997) reported a slightly less even sex ratio in stranded and bycaught animals from Spain, 1.17:1, in this case with more females than males.

Examination of age data from porpoises stranded in the NWIP during 1990–2010 suggests the occurrence of more animals aged 12+ among the strandings towards the end of the study period, implying a decline in mortality rate in porpoises in the NWIP between 2000 and 2010 (Read et al. 2013; see Figure 11A). Read et al. (2013, 2020) used the age-at-death data derived from porpoises stranded in Galicia (Northwest Spain) and north plus central Portugal during 1990–2010 to construct a life table (Table 4A) and to estimate the mortality rate, as well as survivorship and life expectancy at each age (Figure 11B), following the non-parametric Kaplan-Meier approach to estimating survivorship as described in Krebs (1989). The life table also includes calculations for reproductive output, assuming a sex ratio of 1:1 and using estimated values for the proportion of females mature at each age and average pregnancy rate (see 'Life history – Reproductive seasonality, gestation period, pregnancy rate and population reproductive rate'). The estimated annual mortality rate for the entire period was 18%, which is very similar to an estimate for porpoises in Scotland during 1992–2013 (Graham Pierce and Fiona Read, unpublished data). It should be noted that the mortality rate calculated for porpoises in Scotland, also based on stranded animals, may be biased upwards because bottlenose dolphin kills are probably overrepresented in the strandings (since many such mortalities occur close to the coast in northeast Scotland, within the distribution range of the resident bottlenose dolphin population, there is a high likelihood of these animals being stranded). Considering that bottlenose dolphin kills are rarely reported in the Iberian Peninsula, despite the presence of a resident bottlenose dolphin population in Galicia, it is likely that the mortality rate for Iberian porpoises is in fact higher than that in Scotland. The use of the life table approach to estimate mortality rate is, strictly speaking, valid only if population age structure is stable, which can only occur in a stationary population (one in which deaths exactly balance births). Thus, in a declining population (which may or may not be the case here), in which the mortality rate exceeds the birth rate, the age distribution will tend to contract over time with the oldest age classes being lost. Combining data from across many years will then lead to an underestimate of the true mortality rate.

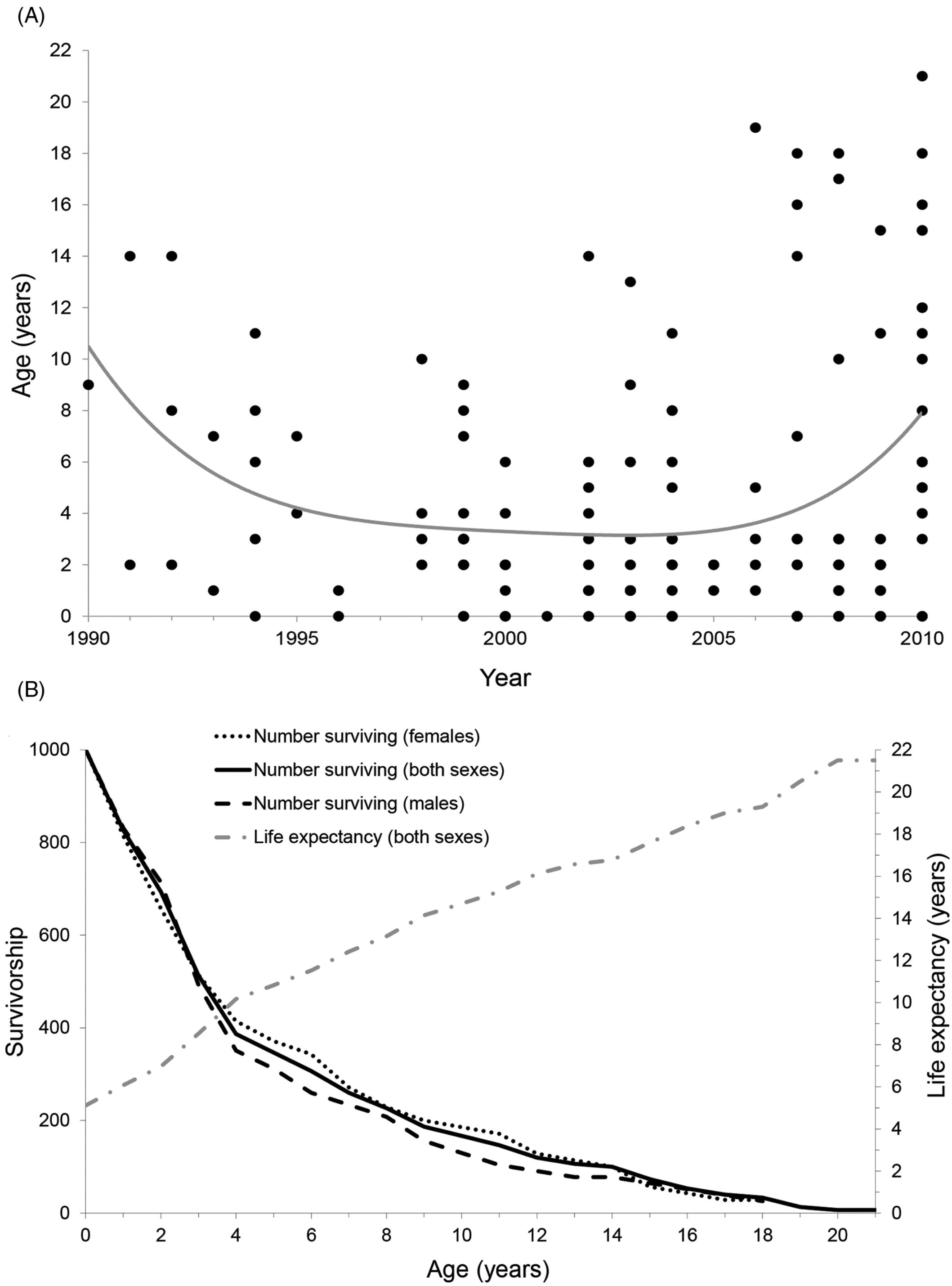

Figure 11 (A) Estimated ages of stranded porpoises from the Northwest Atlantic Iberian Peninsula 1990–2010 (N=151). The fitted smoother represents the underlying trend in average age, which is an indication of the (inverse of the) underlying mortality rate; (B) Estimated survivorship and life expectancy at age for these porpoises.

Table 4A Life Table for Iberian Porpoise, Based on Read et al. (2013) and Following Methodology Described in Krebs (1989)

Age (x)	l_x	d_x	q_x	L_x	T_x	e_x	fem_x	APR =	0.540
								$mat_{f,x}$	b_x
0	151	27	0.179	138	768	5.1	0.5	0	0.0
1	124	20	0.161	114	630	6.1	0.5	0	0.0
2	104	27	0.260	90.5	516	7.0	0.5	0	0.0
3	77	19	0.247	67.5	426	8.5	0.5	0	0.0
4	58	6	0.103	55	358	10.2	0.5	0	0.0
5	52	6	0.115	49	303	10.8	0.5	0	0.0
6	46	7	0.152	42.5	254	11.5	0.5	0.746	9.3
7	39	5	0.128	36.5	212	12.4	0.5	1	10.5
8	34	6	0.176	31	175	13.1	0.5	1	9.2
9	28	3	0.107	26.5	144	14.1	0.5	1	7.6
10	25	3	0.120	23.5	118	14.7	0.5	1	6.8
11	22	4	0.182	20	94	15.3	0.5	1	5.9
12	18	2	0.111	17	74	16.1	0.5	1	4.9
13	16	1	0.063	15.5	57	16.6	0.5	1	4.3
14	15	4	0.267	13	42	16.8	0.5	1	4.1
15	11	3	0.273	9.5	29	17.6	0.5	1	3.0
16	8	2	0.250	7	19	18.4	0.5	1	2.2
17	6	1	0.167	5.5	12	19.0	0.5	1	1.6
18	5	3	0.600	3.5	7	19.3	0.5	1	1.4
19	2	1	0.500	1.5	3	20.5	0.5	1	0.5
20	1	0	0.000	1	2	21.5	0.5	1	0.3
21	1	1		0.5	1		0.5	1	0.3
Sum	843	151							71.6
	M =	0.179						R =	0.085

The table is built from the the number of dead animals recorded (d_x) in each age (year) class (x), from which the notional number of survivors at each age (l_x) can be derived. From these two figures are derived the mortality rate ($q_x = d_x/l_x$)), the average number of animals alive ($L_x = (l_x + l_{x+1})/2$) and total life expectancy ($e_x = \left(\sum_x^{\infty} L_x\right) \Big/ l_x$) for each year class. The overall average annual mortality rate (AMR) is given by $\Sigma d_x / \Sigma l_x$. Assuming a sex ratio (proportion of females fem_x) of 0.5 and a single calf being born to each mother, applying a logistic regression describing the proportion of females that are mature in each age class ($mat_{f,x}$), and the estimated (annual) pregnancy rate in mature females (APR), we estimated the number of births annually per age class ($b_x = l_x \times fem_x \times mat_{f,x} \times APR$). The annual reproductive rate is given by $ARR = \Sigma b_x / \Sigma l_x$.

Table 4B Proportion of Females Mature at Age for Scotland and the Iberian Peninsula

Age	Scotland	Iberia
2	0	0
3	0.18	0
4	0.25	0
5	0.86	0
6	1	0.746
7	1	1

Dividing the study period into four 5-year blocks and comparing survivorship curves for each period, using the non-parametric Kaplan-Meier method as implemented in Minitab (Minitab Ltd), indicated statistically significant variation in mortality rate over time (Read et al. 2013). It should be noted that the overall sample size was 151 animals, so the number of animals per 5-year block was rather low for applying this methodology, and the analysis should be treated as exploratory. Ideally, more age data are needed. To the best of our knowledge, no equivalent analysis is available for other harbour porpoise populations in the Northeast Atlantic. We used a non-parametric bootstrap with 10,000 repeats to estimate the 95% confidence limits of the annual mortality rate, each time sampling with replacement to obtain 151 ages from the original set of 151 ages, then sorted the 10,000 mortality rates and extracted the values from the 251st and 9750th values. The 95% confidence interval for the estimated annual mortality rate of 18% is from 15.3% to 20.5%.

Estimates of mortality rate and survivorship from strandings are potentially subject to various biases. Thus, whether carcasses reach the shore will depend on factors such as the distance from the coast when the animals die, their buoyancy (related to size and body condition) and prevailing currents. Saavedra Penas (2017) found that the youngest age classes of common dolphin were under-represented in strandings data from Galicia (biasing mortality estimates downwards). In addition, mortality in coastal waters will be over-represented compared to mortality in offshore waters, and different causes of death may have different spatial and temporal distributions, hence potentially being over- or under-represented in the subset of animals reaching the shore. In order to address this heterogeneity, a larger sample size (potentially achievable via improved resourcing of some of the strandings monitoring networks in the area and better collaboration between them), the use of drift modelling (supported by carcass release experiments) to estimate both the likely areas of origin of stranded carcasses and the proportion which reach the coast, and better information on the distribution of porpoises at sea, could all be useful.

Reproductive seasonality, gestation period, pregnancy rate and population reproductive rate

The limited evidence available until recently suggested that calving usually takes place in summer in the NWIP. Read (2016) recorded the presence of seven foetuses (in females aged between 6 and 16 years) in Galicia, with the largest (85 cm length) found in July and the smallest (10 cm) in August. Four neonates were found, three in May and one in August, ranging in size from 84.5 cm to 90 cm.

During monthly surveys in southern Galicia by the BDRI between April 2014 and November 2021, calves (animals less than 1 year old) were present in 55 of the porpoise groups sighted (19.2% of the total). The sightings rate for calves (calves seen per hour of effort) increased from winter to autumn (Table 5) (Díaz López & Methion 2018). The marked increase from spring to summer is consistent with most calves being born in summer.

No estimate is available for the duration of gestation period for porpoises in the NWIP but it is presumably within the range reported in the literature; estimates for Europe range from 10 to 12 months (Møhl-Hansen 1954, van Utrecht 1978, Sørensen & Kinze 1994, Bandomir-Krischack

Table 5 Seasonal Distribution of Harbour Porpoise Sightings and Numbers of Adults and Calves

Season	Survey effort (hours)	No of Sightings	No of Adults	No of Calves	SPUE (calves seen/hour)
Winter	235	11	43	2	0.009
Spring	414	42	130	10	0.024
Summer	176	176	832	57	0.077
Autumn	58	58	285	24	0.098
Overall	287	287	1328	93	0.057

Source: Adapted from Díaz López et al. (2022).

1996, Börjesson & Read 2003, Learmonth et al. 2014). The calving interval, during which gestation, lactation and reproductive resting occur, was estimated to be 1.89 years, and females appeared to remain reproductively active until at least 16–18 years old (Read et al. 2013, Read 2016).

Lockyer (2003) stated that pregnancy rates for this species were generally in the range 0.74–0.988. Most of the recent estimates were based on stranded animals and almost all were considerably lower. As noted previously, pregnancy rate tends to be underestimated in samples from strandings although the downward bias can be reduced by considering only trauma deaths and excluding animals stranded during the conception period. As such, pregnancy rate data derived from strandings are difficult to interpret, and it is not clear to what extent the lower rates reported in many more recent studies reflect real changes over time and/or geographical differences and to what extent they are a consequence of the selection of the samples. The annual pregnancy rate (APR; estimated from the proportion of mature females with a foetus between September and May) for Iberian porpoises was 0.54 ($N=13$, Read 2016), which falls in the mid-range of values reported for the North Atlantic. Camarão (2017) reported that 68% of mature females from Portugal (2002–2016) had a corpus luteum in their ovaries but this is also based of a small sample size (although data are presented for a total of 77 females, the analysis of ovarian scars appears to have been based on a subsample of the mature females). The lowest reported values are from North Sea coasts, including 0.34 for the Netherlands (based on the most recent study) and 0.26–0.30 for the English North Sea coast, and the highest was 0.98 in Iceland, where the sample consisted entirely of bycaught animals (Ólafsdóttir et al. 2003); values for the Atlantic coast of North America were also high (0.76–0.93). Learmonth et al. (2014) reported a pregnancy rate of between 0.34 and 0.40 for Scotland, depending on the range of dates selected to reduce error due to missing early term foetuses, but noted that the rate calculated for females from all year round would have been 0.28. Murphy et al. (2015) reported an overall pregnancy rate of 0.29 for the UK (i.e. using combined data from Scotland, England and Wales), noting that this increased to 0.5 among females which had died due to trauma. Murphy et al. (2020) reported an overall pregnancy rate of 0.47 for porpoises from England and Wales, also observing that there were significant regional differences (from 0.29 in the North Sea to 0.6 in the Celtic and Irish Seas). The low pregnancy rate in the North Sea was related to a high proportion of deaths in the infectious disease and "other" (i.e. live stranding, starvation, neoplasia or undetermined) categories, emphasising the relevance of the health status of the sampled animals (and noting that dead animals will almost inevitably include a higher proportion of animals that were in poor health compared to the living population).

Assuming a 1:1 sex ratio and adding data on female maturity (proportion of females mature at each age), plus the estimated pregnancy rate for Iberian porpoises (54%), to the life table for porpoises in the NWIP (Table 4A) results in an overall annual reproductive rate of only 0.085 (assuming that all foetuses survive to full-term), much too low to balance mortality. Even a 100% pregnancy rate in mature females would generate an annual reproductive rate of only 0.157. A pregnancy rate of almost 114% (i.e. with some mature females carrying twins) would be needed to balance the estimated mortality rate. Twins are extremely rare in harbour porpoises. Kompanje et al. (2017) documented a case of conjoined twins and noted that it was only the second known case of twinning in this species (the first having been reported by IJsseldijk et al. 2014). It is true that the estimated pregnancy rate for the NWIP is based on a very small sample of mature females ($N=13$), and, as previously noted, estimates from strandings tend to be strongly biased downwards unless based only on trauma deaths, and the sample size is evidently insufficient to permit a calculation based only on trauma deaths. However, for the population to be maintained, either both the ASM in females or the mortality rate estimates would need to be adjusted. López Fernández (2003) suggested an ASM of 3 years, also based on a small sample of animals. Applying female maturity at age data from a larger sample for Scotland (Table 4B), where 50% of females are mature at 4.35 years of age (Learmonth et al. 2014) to the NWIP population, a pregnancy rate of 0.54 in mature females results in a birth rate of 0.112, still considerably lower than the mortality rate. A pregnancy rate of 86.3% would then be needed to balance the estimated mortality rate (Figure 12).

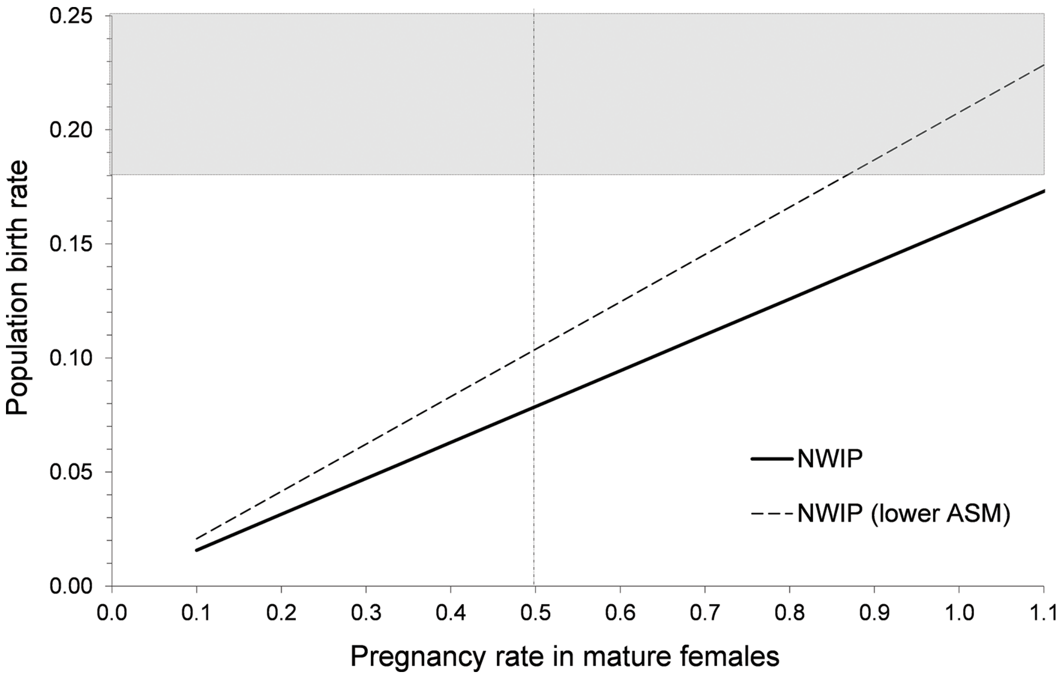

Figure 12 Annual reproductive (birth) rate for the NWIP population as a function of the pregnancy rate in mature females, based on the life table and female age at maturity data from Read et al. (2013). The (grey) shaded area indicates where the annual birth rate equals or exceeds the estimated annual mortality rate. A pregnancy rate exceeding 1.0 would imply that some mature females are carrying twins, which has not been documented in the NWIP. Applying data on Age at Sexual Maturity (ASM) for female porpoises from Scotland from Learmonth et al. (2014) to the NWIP population results in the relationship between pregnancy rate and birth rate indicated by the dashed line.

Feeding ecology

Foraging behaviour

The Iberian porpoise presumably forages mainly on prey distributed along the shelf and continental slopes (i.e. in waters of <200 m depth) since (as mentioned in section 'Distribution and abundance') there are a very few sightings of individuals in the open ocean. The diet of Iberian porpoise has been described based on stomach contents and stable isotope analyses of stranded animals but there have been no specific studies of foraging behaviour.

Harbour porpoises are often described as opportunistic feeders, implying that their prey selection is influenced by availability and not (as optimal foraging theory would predict) by energetic profitability, of which availability is just one component. In practice, we rarely have sufficient knowledge of the factors determining diet selection to justify the use of the word "opportunistic". The apparent large-scale southwards movement of porpoises in the North Sea between 1994 and 2005 may have been due to changes in the distribution and/or availability of prey (Hammond et al. 2013). At a smaller scale, porpoises show seasonal variation in occupancy of foraging grounds (e.g. Nuuttila et al. 2018, Todd et al. 2022), and Williamson et al. (2022) suggested that seasonal shifts in porpoise in distribution in the Moray Firth (northeast Scotland) were related to seasonal prey availability.

Porpoises spend a high proportion of time foraging, "hunting small fish nearly continuously day and night with extreme capture rates" according to Wisniewska et al. (2016), although several studies suggest they are more active at night (e.g. Carlström 2005, V. Todd et al. 2009, Brandt et al. 2014, Osieka et al. 2020, Todd et al. 2022). The field metabolic rate of harbour porpoises may be as much as twice that of similar-sized terrestrial mammals (Rojano-Doñate et al. 2018). Such high energy requirements potentially make them especially susceptible to disturbance and prey depletion. Food consumption, respiration rate and body condition all vary seasonally (e.g. Kastelein et al. 2018), presumably related to both (temperature-dependent) thermoregulatory costs and the energetic demands of the reproductive cycle. Porpoises are more susceptible to disturbance when accumulating body fat before the winter, when their food intake is higher (Gallagher et al. 2021).

Porpoises forage both in the water column for pelagic prey and close to the seafloor for benthic prey. Porpoises have been observed 'bottom grubbing', whereby the animal forages head down, either vertically or at a near-vertical angle, on the seafloor (Desportes et al. 2000, Lockyer et al. 2003). Porpoises use echolocation (biosonar) to help locate and capture their prey, emitting 'foraging buzzes', i.e., click sequences with short inter-click intervals, when hunting (Verfuß et al. 2009). They usually swallow the prey head first, producing suction using their tongue to move it down the throat (Kastelein et al. 1997).

Although often considered to be solitary hunters, porpoises may aggregate in large numbers when foraging in tide race habitats (Pierpoint 2008, Benjamins et al. 2015), and a recent drone-based study in Denmark showed that they may collaboratively hunt in groups of up to six individuals, within which each animal had one or two primary roles, e.g., swimming around the school or herding fish (Torres Ortiz et al. 2021). Díaz López & Methion (2024) observed group sizes of 1 - 46 porpoises in Galician waters, with a mean group size of 5.5, although 62% groups comprised fewer than 5 animals. Porpoises sometimes forage in close proximity to man-made structures such as bridge pillars that attract aggregations of prey (Brandt et al. 2014) as well as around gillnets (Macaulay et al. 2022).

Diet and prey consumption

Harbour porpoise diets in the Northeast Atlantic include species found in all parts of the water column, including demersal and benthopelagic species like European hake *Merluccius merluccius*, Atlantic cod *Gadus morhua* and whiting *Merlangius merlangus* (typically fished with demersal trawl gear), as well as bathypelagic and pelagic-neritic species like blue whiting, sardine (*Sardina pilchardus*), scad (Atlantic horse mackerel, *Trachurus trachurus*) and mackerels (*Scomber* sp.) (typically fished using pelagic gears). Some prey species are relatively large, while others like gobies (Gobiidae), sandeels (Ammodytidae) and bobtail squids (Sepiolidae) (all of which could be considered demersal and spend at least some of the time partially or wholly buried in the substrate of sandy and muddy seabeds) are much smaller. Very often, a few species dominate the diet. Studies of the diet in UK waters from the 1960s and 1970s suggest that small energy-rich pelagic fish like herring (*Clupea harengus*) used to be an important component of the diet (see Santos & Pierce 2003 for a review) but more recent data suggest that it is no longer the case (Santos et al. 2004, G. Pierce Unpubl. data). Of the most important species in harbour porpoise diet in Galicia, blue whiting is generally found on the continental slope, while other important prey, notably pouting *Trisopterus luscus*, scad and European hake, live mainly in shelf waters (see Pierce et al. 2010). Of the minor prey, garfish (*Belone belone*) is pelagic-oceanic, spotted lanternfish (*Myctophum punctatum*) is bathypelagic and argentine (*Argentina* sp.) could be bathydemersal (*Argentina sphyraena*) or bathypelagic (*Argentina silus*) (remains could be identified only to the genus *Argentina*). All are found over the continental shelf but also in much deeper waters, and the occasional presence of remains of such species would be consistent with some foraging occurring in offshore waters.

Spain

Based on the analysis of stomach contents of 72 harbour porpoises stranded on the Galician coast during 1991–2018 (see Table 6), the main prey consumed in terms of biomass were *Trisopterus* spp. (presumably mainly pouting), scad, blue whiting and European hake. The order of importance

Table 6 The Main Prey in the Diet of Harbour Porpoises Stranded in Galicia (1990–2018)

Prey Taxon	%*F*	*N*	%*N*	*W*	%*W*
Fish					
Clupeidae					
Sardina pilchardus	12.5	141	5.4	1113.4	0.9
Gadidae					
Gadiculus argenteus	13.9	278	10.7	1245.5	1.1
Trisopterus spp.	43.1	427	16.5	28,377.3	23.9
Micromesistius poutassou	36.1	517	19.9	22,744.4	19.2
Merlucciidae					
Merluccius merluccius	31.9	140	5.4	18,321.7	15.5
Argentinidae	6.9	21	0.8	415.5	0.4
Sparidae	9.7	13	0.5	588.3	0.5
Gobiidae	22.2	228	8.8	362.7	0.3
Carangidae					
Trachurus trachurus	36.1	209	8.1	23,840.7	20.1
Callionymidae					
Callionymus lyra	9.7	49	1.9	4539.4	3.8
Ammodytidae	5.6	71	2.7	1962.7	1.7
Cephalopod					
Sepiidae					
Sepia officinalis	6.9	12	0.5	410.0	0.4
Sepiolidae					
Sepiola atlantica	18.1	37	1.4	22.0	0.0
Loliginidae					
Alloteuthis spp.	18.1	27	1.0	161.1	0.1
Crustacean					
Decapoda	26.4	–	–	–	–

The importance of each prey taxon is shown as: frequency of occurrence (%*F*), number of prey (*N*), numerical percentage (%*N*), weight of prey (*W*, g) and percentage of reconstructed prey weight (%*W*). Estimated weight of prey is provided in grams (g). Prey taxa with <5% frequency of occurrence are not listed.

of these species changes if the diet is expressed in terms of numbers of prey eaten (blue whiting becomes the most important, followed by *Trisopterus* spp.) or frequency of occurrence (*Trisopterus* spp. remains most important with blue whiting and scad tied for second place). Other important prey included sardine, silvery pout (*Gadiculus argenteus*), gobies and dragonets (Callionymidae). Cephalopods were also present in the diet, especially Atlantic bobtail (*Sepiola atlantica*) and mid-size squid (*Alloteuthis* spp.), but comprised only around 0.9% of prey biomass. Remains of various other invertebrates were found, generally in a small percentage (<5%) of stomachs, although remains of (usually unidentified) decapod crustaceans were found in 26.4% of stomachs. Crustaceans are often assumed to be secondary prey (i.e. from the stomachs of the primary prey) but it is difficult to rule out at least some direct consumption. Several other fish species were found in fewer than 5% of the stomachs, including garfish, Atlantic mackerel *Scomber scombrus* and spotted lanternfish (unpubl. data; see also González et al. 1994, Santos Vázquez 1998, Santos & Pierce 2003, Read et al. 2013, Hernandez-Gonzalez et al. 2024).

Stable isotope data from stranded animals suggest that porpoises in Galicia mainly feed on inshore prey and have a relatively high trophic level, when compared to other common odontocetes in the region, with the exception of coastal bottlenose dolphins (Méndez-Fernandez et al. 2012). Differences in foraging niches between porpoises and other odontocetes inhabiting the NWIP are also revealed by studies that include other ‘ecological tracers’ such as cadmium and PCB concentrations in body tissues (Méndez-Fernandez et al. 2013, 2017).

Inferences on diet may ultimately be possible from studies on macroparasites of the digestive tract but there is so far only one preliminary study for Galicia (Abollo et al. 1998) which examined parasites in four porpoises from Galicia, recording the presence of two species of nematodes, *Anisakis simplex* (which is present in a wide range of fish species) in the digestive tracts and *Halocercus invaginatus* in the lungs. Evidence of trophic transmission of *Halocercus* in cetaceans is circumstantial (based on its absence in unweaned hosts and increased prevalence in older hosts), and in some cetacean species, it appears to be passed from mother to calf (Pool et al. 2021).

Portugal

Aguiar (2013) reported on the diet of harbour porpoise in Portugal based on the examination of stomach contents from 60 harbour porpoises obtained by the stranding network during 1998–2013 (57 from the north, 3 from the south). As in Galicia, the majority of the diet comprised of mainly fish with a small number of cephalopods. Dietary importance was expressed in terms of occurrence and frequency, with common dragonet (*Callionymus lyra*) being the most important species, followed by *Trisopterus* spp. and mullet (*Liza* spp.). Diet composition varied over the years and between the sexes, size classes and cause of death categories (bycaught versus others). Dragonets were markedly more important in the diet of males, porpoises of intermediate size, bycaught individuals and during the period 2006–2009. Bycaught animals might typically be assumed to have eaten prey species targeted by the fisheries but the common dragonet is a benthic inshore fish of no commercial value which is taken as a bycatch in demersal fisheries (e.g. Chang 1951, King et al. 1994), suggesting that bycatch might also arise due to porpoises feeding on non-target species associated with the target catch. A further study based on the stomach contents of 35 porpoises stranded along the Portuguese coast between 2014 and 2016 found that hake, *Trisopterus* spp. and mullet (*Liza* spp.) were the most important prey categories (Pinheiro 2017). The author noted that the diet included pelagic, mesopelagic and demersal species with pelagic species being most important numerically and of similar importance to demersal species in terms of biomass.

Comparing the diet composition for Galicia and Portugal with diets of harbour porpoise populations from other European regions, some differences in main prey can be observed. Fish species of the family Gadidae are often among the most important prey in many areas (e.g. in the Iberian Peninsula, Bay of Biscay, Celtic Sea, North Sea and Baltic Sea; Desportes 1985, Lick 1991, Malinga et al. 1996, Rogan & Berrow 1996, Santos Vázquez 1998, Víkingsson et al. 2003, Santos et al. 2004, De Pierrepont et al. 2005, Spitz et al. 2006, Jansen et al. 2013, Schelling et al. 2014, Andreasen et al. 2017) but the species involved vary regionally. Compared to (say) the diet in Scotland, the importance of pelagic species seems to be greater, and some of the species eaten could have been taken from waters beyond the Continental Shelf.

Santos et al. (2014) used three approaches to estimate food consumption by porpoises in Atlantic waters of the Iberian Peninsula based on data on diet from Galicia, published information on energy requirements and the most recent estimate of population size (SCANS-II 2008). The latter estimate was adjusted *pro rata* considering that the Iberian Peninsula represented approximately 80% of the relevant survey block. It should be noted that the resulting abundance estimate, 1115 animals (95% CI 297–3798), was less than half the abundance estimate given later by Hammond et al. (2013) based on the same dataset and only 38% of the most recent estimate based on the same dataset but corrected for likely negative bias (2880, CV=0.72) by Hammond et al. (2017). The ‘best’ estimate of daily food consumption was 1.96 kg day^{-1}, based on a required daily energy intake of 9292 kJ day^{-1} (Otani et al. 2001), assuming 92.5% assimilation efficiency (Lockyer 2007) and applying the average energy density of prey in the diet of Galician porpoises. This estimate represents approximately 4.4% of the average body weight of stranded porpoises in Galicia, thus falling between values of 3.5% and 4%–9.5% obtained from Yasui and Gaskin (1986) and Kastelein et al. (1997), respectively.

Alternative approaches gave somewhat higher daily food intakes: an estimate of 3.0 kg was obtained based on Kleiber (1947), assuming an active metabolic rate three times the basal metabolic rate and again applying an assimilation efficiency of 92.5%. Applying the Innes et al. (1987) equation for food consumption as a function of body weight to estimated body weights of stranded porpoises, resulted in an estimated 3.45 kg of food eaten daily by an average porpoise. The annual estimates for consumption of fish by the porpoise population, assuming that the diet in Galicia could be applied to the whole area, were 39 t (95% CI 2–251 t) of clupeids, 258 t (95% CI 61–1015) of gadids, 72 t (95% CI 11–333 t) of hake and 140 t (32–658 t) of scad. Scaling up to reflect the most recent abundance calculation based on SCANS II data, the annual consumption figures would be 101, 666, 186 and 362 tonnes, respectively.

Threats to the Iberian harbour porpoise

Given the low population size of Iberian porpoise and the importance of the region for fisheries, fishery bycatch is a relevant concern, the more so given the evidence from strandings of significant bycatch mortality, already apparent in the 1980s, as mentioned by Sequeira and Ferreira (1994), who also suggested that there had been an associated decline in abundance. There was a marked decline in genetic diversity between 1990-2002 and 2012-2015 (Ben Chehida et al. 2023), consistent with a substantial decline in effective population size, although the similarity of abundance estimates from the large-scale surveys in 2005 (SCANS-II) and 2016 (SCANS-III), and the apparent increase in abundance seen in the 2022 (SCANS I IV) survey, suggest that abundance was stable or increasing since 2005 (Hammond et al. 2017, 2021, Gilles et al. 2023).

Other relevant anthropogenic threats include contaminants (notably Persistent Organic Pollutants (POPs)), pathogens (including parasites), underwater noise and other forms of disturbance, and collisions with boats. Going back in time, hunting can be added to this list. In relation to natural threats, porpoises may be predated (e.g. by killer whales) and fatally attacked by bottlenose dolphins. In relation to anthropogenic threats, porpoises in the Iberian Peninsula are protected by several European directives, notably the EU Habitats Directive and MSFD, as well as EU fishery regulations and national laws. As such, the status of porpoise populations is routinely monitored and assessed, and, depending on the outcomes of these assessments, mitigation action may be legally required (see section 'Conservation of porpoises in Europe: legal Protection and its implementation' for details).

Fishery bycatch

Fishery bycatch mortality has long been recognised as one of the most severe threats facing cetacean populations worldwide (IWC 1994). It is the proximate cause of the perilous state of a number of populations, including the vaquita in the Gulf of California and the Baltic Proper subpopulation of harbour porpoise. Bycatch mortality of harbour porpoise has been a cause for concern in many parts of the species' range (Gaskin 1984, Read & Gaskin 1988, Kirkwood et al. 1997, Tregenza et al. 1997, Caswell et al. 1998, Cox et al. 1998). In Europe, porpoise bycatch has been a cause for concern for more than two decades – and bycatch in some fisheries was and/or is alarmingly high. For example, based on observer data, Vinther (1999) reported that annual porpoise bycatch mortality in Danish gillnets in the North Sea was around 7000 animals per year. By 2001, estimated annual bycatch mortality had fallen to between approximately 2900 and 3900, depending on the method used to extrapolate from observed bycatch to total bycatch (Vinther & Larsen 2004), and the most recent estimate (for 2020) was from approximately 2100 to 2600, depending on whether the expected impact of pinger deployment was taken into account (Kindt-Larsen et al. 2023).

In Europe, fishery bycatch mortality of porpoises is monitored under Regulation 2019/1241 (and was previously monitored under Regulation 812/2004) and is assessed (nationally) under the MSFD. In their MSFD assessments of porpoise, both Spain and Portugal assessed its status as "Not good" in relation to bycatch mortality (Anon. 2022b). According to the most recent Indicator

Assessment by OSPAR, bycatch of porpoises in three Assessment Units (Greater North Sea, Irish and Celtic Seas, and West of Scotland and Ireland) exceeded the threshold, while despite the (apparent) absence of records of porpoise bycatch in the Iberian Peninsula during 2015–2020 from the on-board observer programme, other sources of evidence suggest that bycatch is well above the threshold (Taylor et al. 2022).

The small size of the Iberian porpoise population means that it has a limited capacity to withstand high bycatch mortality. Recent evidence of cetacean bycatch mortality in Iberian Peninsula waters has emerged from two main sources, on-board observations and the examination of stranded animals. Some additional information is available from other sources, e.g., interviews with fishers (e.g. López et al. 2003, Vingada et al. 2011, Goetz et al. 2014, Martínez-Cedeira & López 2018) and historical records. Until the late twentieth century, human consumption of small cetaceans, most likely of bycaught animals, was common in both Portugal and Spain (Sequeira 1996, López et al. 2003).

Observer, stranding and interview-based estimates of bycatch mortality are all subject to limitations and biases, which should be borne in mind when reading the remainder of this section. It is notable that the coverage provided by observer data tends to be both low and very patchy: data for some fleets were reported only in some years, and small-scale vessels were often entirely excluded. On the other hand, only observer data can provide reliable and unambiguous information on the precise time and location of the bycatch and the gear involved, and they are more likely to have been collected following a specified sampling strategy. Especially in the case of interviews, there may be doubts about the species involved, and about the accuracy and veracity of information that is provided by interviewee. In the case of strandings, the origin and representativeness of the carcasses examined may be in doubt.

For all data sources, the way in which data are processed and raised to fleet or population level is also relevant. For example, in the case of strandings, it is important to know whether undiagnosed deaths were excluded or considered as non-bycatch, and how “possible” or “probable” bycatch mortalities were treated. When observer data have been used, it is often the case that for fleets where observer coverage was very low, any observed bycatches from these fleets were not included in the estimates of total bycatch.

On-board observations

Since 2011, records of cetacean bycatch recorded by observers on vessels fishing in EU Atlantic waters have been compiled annually by the ICES Working Group on Bycatch of Protected Species (WGBYC) in order to permit estimation of total bycatch mortality. While EU Regulation 812/2004 was in force, these data were collected mainly by dedicated observers (although not by France or Spain) but nowadays, under Regulation 2019/1241, they are mainly from fisheries observers (who had, however, previously provided records of bycatch of other protected, endangered and threatened species (PETS)). For the Iberian porpoise, the most relevant ICES fishery divisions are 8c (northern Spanish Atlantic coast) and 9a (western Spanish Atlantic coast, Portuguese coast and southern Spanish Atlantic coast).

Spain Fernández-Contreras et al. (2010) carried out observations on-board pair trawlers in Galicia in 2001 and 2002 and estimated an annual bycatch of 394 common dolphins but no harbour porpoises. The only monitoring of cetacean bycatch in Spain by dedicated on-board observers under Regulation 812/2004 was during a 12-month pilot project (2008–2009) in ICES Subarea 8 (Anon. 2009b, Lens & Díaz 2009). Coverage in 2008 included only the last quarter of the year and recorded no porpoise bycatch. Thirteen porpoises were bycaught in 2009, leading to an estimated total of 208 bycatch mortalities of porpoises in gillnets over the course of the study. However, all the porpoise bycatch occurred further north in the Bay of Biscay, in ICES Divisions 8a and 8b, and thus outside Iberian waters. Based on the data collected by fishery observers, Spain reported bycatches of protected species in pelagic trawls in Division 8c in 2017, for bottom trawls, nets and pelagic trawls in Division 8c in 2018, for rods and lines in Division 8c and Subdivision 8d2 in 2019, and for bottom trawls and nets in 8c and 8d2 in 2020. In all these years, there was no reported bycatch of harbour porpoise (ICES 2019, 2020c, 2021c).

Portugal Information on fishery bycatches of porpoises in Portugal published in papers and project reports is summarised in Table 7. During July–October 2003, cetacean sightings and bycatch interactions were recorded during on-board monitoring of 48 fishing trips by purse-seine vessels operating from four Portuguese ports (Figueira da Foz, Sesimbra, Setúbal and Sines), and interviews with fishers were conducted during 36 trips. No porpoise bycatch was recorded but porpoises were sighted once (among 31 cetacean sightings), with two individuals being seen near Figueira da Foz (Wise et al. 2007). Monitoring of 292 beach seine (*xávega*) hauls in Portugal between 2008 and 2011 recorded five porpoise mortalities, i.e., a mortality rate of 0.017 animals per haul (Vingada et al. 2011). The authors stated that they observed 3.3% of national fishing activity in this fleet. However, based on broad consistency with annual fishing effort data reported in Oliveira et al. (2015), it appears that 292 hauls represented approximately 3.3% of *annual* fishing effort by the fleet. Thus, the annual number of beach seine hauls would be around 8850, implying a total annual bycatch mortality in this gear of around 150 porpoises. Vingada and Eira (2018) reported results on bycatch mortality from the MARPRO project, indicating that the estimated average annual porpoise bycatch during 2010–2015 comprised 17 animals in purse seines, 203 in the polyvalent fleet, 0 in bottom trawls and long lines and 21 in beach seines. This leads to an annual bycatch estimate of 241 porpoises per year. Marçalo et al. (2015) reported on observations on-board purse seiners in Portuguese waters in 2010–2011 (163 days at sea) which included one porpoise bycatch, an animal which was observed being encircled and subsequently escaping. This record was included within Portugal's submission to the 2011 meeting of the ICES Working Group of Bycatch (A. Marçalo, Pers. Obs.). Although such observations may be rare, this illustrates the point that not all recorded cetacean bycatches lead to immediate mortality.

According to reports submitted to the ICES Study Group on Bycatch of Protected Species (SGBYC) in 2010 and to ICES WGBYC in 2011 (ICES 2010, 2011), in 2007–2009, Portugal had no fisheries covered by EU Regulation 812/2004 (see section 'Conservation of porpoises in Europe: legal Protection and its implementation') and no observer programme. In 2010, Portugal reported on observations of polyvalent and purse seine fisheries in Division 9a, reporting five bycaught porpoises in the polyvalent fishery. Observer data for 2010 from the polyvalent fleet data were reported separately for boats targeting demersal and pelagic fish. Porpoise bycatches came from boats targeting mackerel, horse mackerel and sardine. Scaling up (based on the proportion of effort observed in these fisheries), this leads to an estimate of 150 porpoises bycaught (ICES 2013a). In 2011, Portugal reported on observations of purse seine, demersal and polyvalent trawl fisheries, with one reported porpoise bycatch in the purse seine fishery, leading to an estimate of 103 porpoises bycaught

Table 7 Published Information on Harbour Porpoise Bycatch in Portuguese Waters (ICES Division 9a) Collected by On-board Observers during 2003–2020

(A) Data published in scientific papers and reports (for the years 2003–2015)

Years	Country and Source	Fishing Area	Métier	Observed/ Total Effort (% Coverage)	No. of Incidents	No. of Specimens Bycaught	Extrapolated Annual Bycatch Estimate	Effort Unit
2003	Portugal (1)	9	Purse seine	48	0	0	0	Trips
2008–2011	Portugal (2)	9	Beach seine	292 (3% of annual total) [a]		5	150	Hauls
2010–2011	Portugal (3)	9	Purse seine	163	0	0	0	Days
2010–2015	Portugal (4)	9	All gears				241	

(*Continued*)

Table 7 (*Continued*) Published Information on Harbour Porpoise Bycatch in Portuguese Waters (ICES Division 9a) Collected by On-board Observers during 2003–2020

(B) Data extracted from ICES WGBYC reports from 2011 to 2021 for individual year and gear combinations during 2010–2020)

Year	Country and Source	Fishing Area	Vessel Size (m)	Métier	Target Species	Season	Observed/ Total Effort (% Coverage)	No. of Incidents	No. of Specimens Bycaught	Extrapolated Annual Bycatch (with 95% CI)	Effort Unit
2010	Portugal (5)	9a	NA	Polyvalent	*Trachurus* spp, sardine, chub mackerel	May–October	80/2400 (3.33%)	NA	5	150[b] (23–277)	NA
2011	Portugal (6)	9a	>15	Purse-seine	NA	NA	110/11,320 (0.97%)	NA	1	[103][c] (0–304)	NA
2012	Portugal (7)	9a	>12	Polyvalent	Mixed demersal	NA	71/63,612 (0.11%)	1	1	896[d] (0–2640)	Haul
2015	Portugal (8)	9a	>12	Polyvalent	NA	NA	245/59,679 (0.41%)	NA	6	1462[d] (306–2617)	Days at sea
2010–2015	Portugal (5–8)	9a	>12	Sum across all monitored gears				NA	13	418[e] (86–922)	
2010–2020	Portugal (5–8)	9a	>12	Sum across all monitored gears				NA	13	228[e] (47–503)	

Sources: Years and gear combinations with no reported bycatch of porpoises or no reported observer data are not included in the table. Combined estimates for all monitored gears are given for 2010–2015 and 2010–2020 (note that no bycatch of porpoise was reported for this area during 2016–2020. Approximate 95% confidence limits based on the normal approximation method are provided. Multiannual estimates and 95% confidence limits were obtained summing annual values and dividing by the number of years. Data sources:(1) Wise et al. (2007), (2) Vingada et al. (2011), (3) Marcalo et al. (2015), (4) Vingada and Eira (2018), (5) ICES (2013a), (6) ICES (2013b), (7) ICES (2014c), (8) ICES (2017).

[a] The source text implies that it is 3.3% of total fishing effort over 4 years but based on the reported amount of fishing effort for this metier at this time, it appears to be 3.3% of the annual effort. If this latter interpretation is incorrect, the annual bycatch would be 37.5.

[b] Portugal reported an extrapolated bycatch of 80 in its annual Reg. 812/2004 report, possibly because it separated observations of demersal and pelagic fishing by the polyvalent fleet, wth porpoises being caught only during the latter.

[c] Although this was a bycatch, Marcalo et al. (2015) commented the animal escaped from the net. It should thus not be counted as bycatch mortality.

[d] ICES WGBYC did not extrapolate from these data to estimate bycatch.

[e] The average values over 2010–2015 (6 years) and 2010–2020 (11 years) were calculated excluding the animal caught in 2011 because it was not a bycatch *mortality*.

(ICES 2013b). As noted above, according to Marçalo et al. (2015), the animal escaped from the net and should thus not be counted as bycatch mortality. In 2012, Portugal reported observations from demersal trawl, purse seine and polyvalent (trammel net) fishing to ICES WGBYC (ICES 2014c). Bycatch of one porpoise was recorded in the polyvalent fleet deploying trammel nets. With 63,612 effort units (probably individual hauls), of which 71 were observed, the extrapolated bycatch would be 896 porpoises. In 2013 and 2014, Portugal reported on cetacean bycatch in polyvalent, seine and bottom trawl fleets, which did not include any porpoises (ICES 2015, 2016). In 2015, only observations from the polyvalent fleet were reported, with a bycatch of six porpoises in set nets (ICES 2017). Extrapolation would give a porpoise bycatch of 1462 animals. A small number of trips by boats in the polyvalent fleet deploying fixed nets was observed in 2016, with no porpoise bycatch (ICES 2018a). In 2017, Portugal reported zero bycatch of harbour porpoises in bottom trawls, nets and seines. In 2018, reporting covered bottom trawls, nets, seines and surrounding nets but again there was no observed bycatch of harbour porpoise (ICES 2020c). In 2019 and 2020, no porpoise bycatch was recorded for longlines, nets and surrounding nets (ICES 2021c). See Table 7(B) for a summary of harbour porpoise bycatch in Division 9a reported by ICES WGBYC.

Based on the Portuguese observer data for 2010–2015 compiled in Table 7B, assuming that all the fishing took place in Portuguese waters and that bycatch in beach seines took place only in the years during which that fleet was observed (2008–2011), the estimated average annual bycatch during 2010–2015 across polyvalent and purse seine fleets (435, or 418 if the porpoise which escaped the purse seine in 2011 is not included in the calculations), plus the bycatches in beach seines (50 per year over 6 years assuming that 150 animals were taken in both 2010 and 2011), would be 485 or 468 animals. Taking the absence of observed bycatches over the following 4 years at face value, the annual average for 2010–2020 (including an average of 300/11 = 27 animals taken in beach seines per year is 264 or 255 porpoises per year. Total bycatch is likely to be underestimated because observer effort focused on larger boats (>12 m), and fishing by Spanish vessels is evidently not included. On the other hand, we have included figures that ICES WGBYC did not use: they did not include project-based data collection and did not extrapolate from the bycatches recorded in 2012 and 2015, presumably due to the very low percentage of effort that was monitored. The 95% confidence intervals for the annual bycatch estimates are also provided in Table 7B, based on the normal approximation method to derive 95% confidence limits for a binomial variable, assuming that a single porpoise was caught during each bycatch event (this information was not always available in the WGBYC reports) and thus treating the number of bycaught animals divided by the number of observed fishing events as a binomial variable. If the lower 95% limit was below zero or the upper limit above 100%, we give the values as 0% and 100%, respectively. The estimates and confidence intervals for bycatch per observed fishing event were extrapolated to give the number of animals bycaught annually, by dividing by the proportion of annual fishing events that were observed. Multiannual estimates and associated 95% confidence limits were obtained summing annual values and dividing by the number of years.

The ICES Workshop on estimation of MOrtality of Marine MAmmals due to Bycatch WKMOMA (ICES 2021c) estimated bycatch rates for marine mammals in the OSPAR area using submitted bycatch data for 2005–2021. For the Iberian coast, these data contained only a single porpoise bycatch record, from trammel nets targeting crustaceans (gear code GTR CRU) and no records of bycatch during 2015–2020. Due to the lack of data, it was not possible to assess bycatch for the Iberian coast. Evidently there was a discrepancy with previously reported data, which included several porpoise bycatch events, including six animals bycaught in 2015.

Some of the estimates of bycatch mortality from the on-board observer data are extremely high. In all cases, the extrapolation was based on a small number of observed bycatches, the proportion of fleet activity observed was low and the estimated 95% confidence intervals are very wide. Where the proportion of effort was very low, ICES WGBYC did not extrapolate from these values. If the bycatch mortality is as high as these estimates suggest, there is an apparent contradiction with the lack of change in population size seen between the 2005 and 2016 SCANS II and III surveys. As will be seen below, this issue is not unique to the observer data on bycatch.

Evidence from strandings

Spain Information on bycatch mortality diagnosed from stranded animals in northern Spain mainly derives from the Galician stranding network (CEMMA), since (to date) the other stranding networks operating in northern Spain rarely report on diagnosed bycatch mortality. Read et al. (2013, 2020) compiled data for 1990–2010. Among the 313 (i.e. approximately 15 per year over the whole period) stranded and (a small number of known bycatches) handed in by fishers, 23% were known or diagnosed bycatches, 17% showed no evidence of bycatch, 11% were examined but the cause of death could not be determined and 47% were not examined due to being too decomposed. Based on these data, we excluded the last category from analysis and calculated the % bycatch mortality among "strandings" both with and without the known bycatches handed in by fishers (Table 8). Data for 1990–2019 were summarised by Pierce et al. (2020): there were 306 strandings of harbour porpoise, i.e., on average around 10 per year, 53 of which (17.3% of the total) showed evidence of fisheries interactions. Of these animals, 120 were in a good state of preservation (decomposition state of 3 or less) of which 41 (34.2%) showed evidence of bycatch. Known bycatches ($N=7$) handed in by fishers were excluded from the calculations. These compilations of data result in similar estimated percentages of stranded animals dying due to bycatch, 31.0% and 34.2%, respectively, when known bycatches are excluded; when known bycatches were included, the former estimate increased to 34.8%, i.e., an upward bias of almost 4%. As for the observer-based bycatch estimates, confidence

Table 8 Strandings of Harbour Porpoises in Galicia 1990–2021: Incidence of Bycatch Mortality

Period, Subset and Source	No. of Strandings	No. F + SD Carcasses Examined	No. with Bycatch Evidence	No. with Bycatch Evidence per Year	No. Bycaught/No. Examined (%)	95% CI for % Bycaught (%)
1990–2010 (no KB) (1)	208	87	27	1.3	31.0	21.3–40.8
1990–2010 (with KB) (1)	213	92	32	1.5	34.8	25.1–44.5
1990–2019 (2)	306	120	41	1.4	34.2	25.7–42.7
1990–1999 (3)	105	38	14	1.4	36.8	21.5–52.2
2000–2009 (3)	95	20	9	0.9	45.0	23.2–66.8
2010–2021 (3)	152	67	27	2.3	40.3	28.6–52.0
1990–2021 (3)	352	125	50	1.6	40.0	31.4–48.6
2018–2020 (4)	51	18	8	2.7	44	21.5–67.4
2000–2020 (PB→NB) (5)	213	83	34	1.6	41.0	30.4–51.5
2000–2020 (PB→B) (5)	213	83	40	1.9	48.2	37.4–58.9

Number of stranded porpoises, number of fresh and slightly decomposed (F + SD) carcasses examined, number with bycatch evidence, number with bycatch evidence per year, and the percentage with bycatch evidence (with 95% confidence intervals estimated using the normal approximation method). All data were collected by CEMMA. For the 1990–2010 data we present results both without and with known bycaught animals (KB, i.e. animals handed in by fishers) included. The 1990–2019 data exclude the known bycaught (KB) animals. For the 2000–2020 data we present results for both (1) treating probable bycatch (PB) as non-bycatch (NB) and (2) treating probable bycatch as bycatch (B)

Sources: (1) Read et al. (2013, 2020), (2) Pierce et al. (2020), (3) CEMMA, unpublished, (4) ICES (2020c, 2021c), (5) Torres-Pereira et al. (2023a).

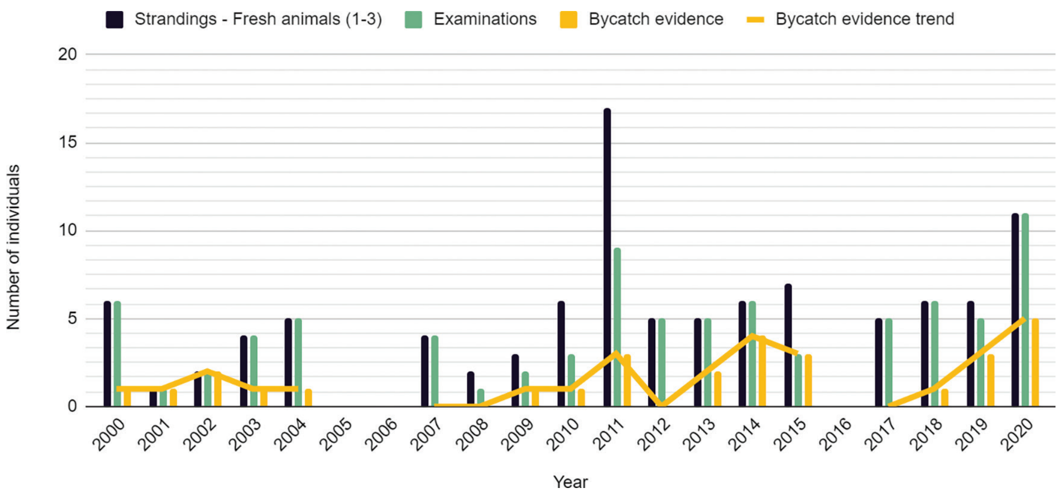

Figure 13 Number of fresh or slightly decomposed harbour porpoises stranded, number of examinations and number of examined stranded animals with evidence of fishery bycatch from 2000 to 2020 in Galicia (NW Spain). In 2005 and in 2006, none of the harbour porpoise strandings was registered as fresh or slightly decomposed. There were no data available for 2016. The trend line refers to the number of bycaught animals recorded. (Data provided by CEMMA.)

limits around the percentage of stranded animals killed by bycatch can be derived using the normal approximation method. Even combining data over two or three decades, the lower and upper confidence intervals span a range of almost 20% (Table 8). Multiplying the 34.2% of strandings that were bycaught by the estimated annual mortality rate of 18% (Read et al. 2013, 2020) gives the annual bycatch mortality rate of 6.2%. Given a population size estimate of 2900 animals, this equates to 178 annual deaths due to bycatch. Finally, the estimate of 34.2% was based on 41 bycaught animals over 30 years, giving a minimum estimate of average bycatch mortality per year of 1.4 animals.

Updated and revised numbers for 1990–2021 provided by CEMMA and a summary for 2018–2020 taken from ICES WGBYC reports (ICES 2020c, 2021c; again based on CEMMA data) also appear in Table 8. The overall percentage of mortality attributed to bycatch across approximately three decades was 40%. The average number of stranded porpoises recorded annually was highest in the most recent period (2010–2021) (12.7) as was the average number of bycaught animals recorded per year (2.3 on average, see also Figure 13). During 1990–2021, the estimated percentage of bycatch mortalities in the strandings was 40% (95% CI 31.4%–48.6%).

Torres-Pereira et al. (2023a) summarised porpoise strandings data from Galicia for the period 2000–2020 (alongside Portuguese data), distinguishing certain and probable bycatch mortalities. In the calculations presented in Table 8, we firstly used only (certain) bycatch and considered probable bycatches as non-bycatch and secondly counted certain and probable bycatches as bycatch. Following the first approach, 41.0% of stranded animals were bycatches, rising to 48.2% when probable bycatches were included.

Portugal As noted previously, since strandings monitoring began in 1977 in Portugal, a decline in strandings was seen up until the late 1980s. Many of the animals showed evidence of bycatch mortality, such as net marks around the head and flippers, presumably due to entanglement in fixed nets (Sequeira & Ferreira 1994). A summary of strandings data from mainland Portugal between 1979 and 2009 suggested that approximately 24% of porpoises strandings were known bycatches or showed indications of bycatch, but no detail was provided on changes over time (Gonçalves de Sousa 2010). Between 2000 and 2016, 4021 stranded cetaceans were reported from the Portuguese coast. Cause of death was determined for 2423 animals, and bycatch was the most frequently diagnosed cause of mortality (44.3%) (Vingada & Eira 2018). Although the report did not provide a full account for each species, it was stated that, of 43 porpoises recorded as stranded in 2014, the most frequent cause of death was bycatch (64%). The southern strandings network recorded 25 porpoise strandings along the south coast of Portugal between 2010 and 2017. Cause of death was recorded in seven cases and two of these were bycatch mortalities (A. Marçalo, Pers. Obs.; see Table 9). Ângelo (2020) compiled strandings data for the west coast of Portugal for 2010–2019, referring to 130 bycaught and probably bycaught animals among 278 strandings. Most bycatches were recorded in spring (54) and summer (37). Based on marks that could be used to identify the gear responsible, bycatches in spring were divided almost equally between those caught in fixed nets (28) and those taken by other and undetermined gears (26). Of the latter, more than half were due to beach seine. In summer, a majority (28) of the bycatches were in the other and undetermined gears category.

Aguiar (2013) reported on the diet of harbour porpoises in Portugal based on the examination of stomach contents from 60 harbour porpoises obtained by the stranding networks during 1998–2013 (57 from the north, 3 from the south). Of these animals, 37 were known or diagnosed bycatch mortalities

Table 9 Strandings of Harbour Porpoises in Portugal 1998–2020: Incidence of Bycatch Mortality

Year, Region, Source, Subset	No. of Strandings	No. F + SD Carcasses Examined	No. with Bycatch Evidence	No. with Bycatch Evidence per Year	No. Bycaught/No. Examined (%)	95% CI for % Bycaught (%)
1990–2010 North (no KB) (1)	97	72	44	2.1	61.1	49.9–72.4
1990–2010 North (with KB) (1)	100	75	47	2.2	62.7	51.7–73.6
1998–2013 (2)	130	60	37	2.3	61.7	49.4–74.0
2014 (3)	43	–	–	–	64	–
2010–2017 South (4)	25	7	2	0.3	28.6	0–62.0
2010–2019 North (7)	278	–	130	13	–	–
2000–2020 North (5) (PB→NB)	453	264	136	6.5	51.5	45.5–57.5
2000–2020 South (5) (PB→NB)	71	17	1	0.0	5.9	5.3–17.1
2000–2020 all (5) (PB→NB)	524	281	137	6.5	48.8	42.9–54.6
2000–2020 North (5) (PB→*B*)	453	264	168	8.0	63.6	57.8–69.4
2000–2020 South (5) (PB→*B*)	71	17	3	0.1	17.6	0–35.8
2000–2020 all (5) (PB*v*B)	524	281	171	8.1	60.9	55.1–66.6
2011–2015 all (5) (PB→NB)		96	49	9.8	51.0	41.0–61.0
2018–2020 (6)	118	72	28	9.1	38.9	27.6–50.1

Number of stranded porpoises, number of fresh and slightly decomposed (F + SD) carcasses examined, number with bycatch evidence, and the percentage with bycatch evidence (with 95% confidence intervals estimated using the normal approximation method). For the 1990–2010 data we present results both without and with known bycaught animals (KB, i.e. animals handed in by fishers) included. For the 2000–2020 data we present results for both (1) treating probable bycatch (PB) as non-bycatch (NB) and (2) treating probable bycatch as bycatch (B).

Sources: (1) Read et al. (2013, 2020), (2) Aguiar (2013) and Vingada and Eira (2018), (3) Vingada and Eira (2018), (4) Ana Marçalo (Pers. Comm.), (5) Torres-Pereira et al. (2023a), (6) ICES (2020c, 2021c), (7) Angelo (2020).

and a further nine were probable bycatch mortalities. Cause of death was undetermined in ten animals. It is not clear if this last category refers to animals in a good state of preservation for which cause of death could not be determined or animals which were not fresh so that diagnosis was not possible. We have assumed the former. Thus, the minimum estimate for the percentage bycatch mortality (assuming the undetermined deaths were not bycatch) would be 61.7% and the maximum (including probable bycatches) would be 76.7%. Since the sample included an unspecified (albeit probably small) number of animals handed in by fishers, the estimated percentage of bycaught animals is likely biased upwards. During the 16-year period covered by the dietary study, a total of 130 porpoises was recorded as stranded in Portugal (Vingada & Eira 2018). Based on the proportion of bycaught animals reported by Aguiar (2013), the total number of porpoise bycatch mortalities recorded by the network during this period would have been between 61.7% and 76.7% of 130, i.e., between 80 and 100 porpoises. If these figures are scaled up to the whole Iberian population, assuming annual mortality of 18% and a population of 2900 animals, the overall annual bycatch mortality rate would be between 11.1% and 13.8%, and the annual bycatch mortality in this population would thus be between 322 and 400 animals.

Torres-Pereira et al. (2023a) presented results from the period 2000–2020, also separately presenting results for 2011–2015 (years for which there was an annual abundance survey). They estimated an average annual bycatch mortality in Portuguese waters of 207 porpoises per year during 2011–2015. They provided separate estimates for north (plus central) and southern Portugal. Based on their data, the estimated bycatch rate among strandings during 2000–2020 was much lower in the south (5.9% or 17.6% depending on whether "probable" bycatches are rated as bycatch or not) than in the north (51.5% or 63.6%), with the combined figures (48.8% or 60.9%) being closer to the latter. Information on numbers of stranded harbour porpoises with evidence of bycatch in Portugal during 2018–2020, as submitted to ICES WGBYC (ICES 2020c, 2021c), suggested a lower percentage of bycaught animals (38.9%) among strandings than most of the estimates from earlier years. The 95% confidence intervals for most of the estimates are wide (spanning 20%+), although narrower (10%–12%) for the large 20-year data sets from the north and all Portugal. Applying the 18% annual mortality rate and population size of 2900, to higher (60.9%) and lower (38.9%) estimates of the percentage of bycatches in Portuguese strandings gives annual population bycatch mortality rates of 10.4% and 7.0%, respectively, equivalent to 303 or 203 animals.

Combined data for Portugal and Galicia As described in section 'Life history – Cause of death, sex ratio, age structure and mortality rate', Read et al. (2013, 2020) used life history data derived from porpoises stranded in Galicia (Northwest Spain) and north-central Portugal during 1990–2010 (plus a few known bycatches handed in by fishers) to estimate the annual mortality rate (18%). Note, however, that the current strandings network in northern Portugal started only in 2010 so the first decade of data are from Galicia alone. These authors then estimated how much of this annual mortality was due to bycatch, based on the proportion of these animals that showed signs of bycatch. Among the 313 stranded and handed-in bycaught porpoises during 1990–2010, 23% were known or diagnosed bycatches, 17% showed no evidence of bycatch, 11% were examined but the cause of death could not be determined and 47% were not examined due to being too decomposed. Evidently, as previously mentioned, including even a small number of animals handed in by fishers will likely result in an upward bias in the estimated bycatch rate. We calculated the percentage of bycaught individuals among all those that were examined as well as repeating the calculations excluding known (handed-in) bycatches. The resulting percentages for the combined dataset were 47.3% (all animals examined) and 44.7% (excluding known bycatches). The 95% confidence intervals span a range of approximately 15% (Table 10). These figures are equivalent to annual bycatch mortality rates in the population of 8.5% (247 deaths annually) or 8.0% (233 deaths annually).

Torres-Pereira et al. (2023a) summarised porpoise strandings data from north Portugal, south Portugal and Galicia for the period 2000–2020. During this period, 756 porpoise strandings were logged. The authors noted that the number stranded per kilometre of coastline in Portugal

Table 10 Strandings of Harbour Porpoises in Galicia and Portugal Combined, 1990–2020: Incidence of Bycatch Mortality

Years, Area, Regions, Subset, Source	No. of Strandings	No. F+SD Carcasses Examined	No. with Bycatch Evidence	No. with Bycatch Evidence per Year	No. Bycaught/ No. Examined (%)	95% CI for % Bycaught (%)
1990–2010, Galicia+North Portugal, no KB (1)	305	159	71	3.4	44.7	36.9–52.4
1990–2010, Galicia+North Portugal, with KB (1)	313	167	79	3.8	47.3	39.7–54.9
2000–2020, Galicia+all Portugal (PB→NB) (2)	756	364	171	8.1	47.0	41.9–52.1
2000–2020, Galicia+all Portugal (PB→*B*) (2)	756	364	211	10.0	58.0	52.9–63.0

Number of stranded porpoises, number of fresh and slightly decomposed (F + SD) carcasses examined, number with bycatch evidence, and the percentage with bycatch evidence (with 95% confidence intervals estimated using the normal approximation method). For the 1990–2010 data we present results both without and with known bycaught animals (KB, i.e. animals handed in by fishers) included. For the 2000–2020 data we present results for both (1) treating probable bycatch (PB) as non-bycatch (NB) and (2) treating probable bycatch as bycatch (B).

Sources: (1) Read et al. (2020), (2) Torres-Pereira et al. (2023a).

was more than double that in Galicia. The animals were examined to determine the cause of death in 364 cases, of which 171 (47.0%) were diagnosed as bycatch mortalities and a further 40 (11.0%) as probable bycatches. Animals for which no cause of death could be determined but which did not present signs of bycatch were considered as non-bycatch. Breaking this down by area, and considering only certain bycatches, the percentage of bycatch mortality was highest in north Portugal (51.5%), somewhat lower in Galicia (41.0%) and lowest (albeit based on a much smaller sample size) in south Portugal (5.9%) (Tables 8 and 9). Combining data from all three networks gives an overall percentage of bycatch mortality among strandings of 47.0%. The 95% confidence intervals for these estimates span a range of approximately 10% (Table 10).

Applying the previously mentioned 18% annual mortality rate and population size of 2900, this translates into an annual population mortality rate due to bycatch of 8.5% or 245 animals per year. If we count probable bycatches as bycatch, these figures increase to 58.0% bycatches among the stranded animals, 10.4% annual mortality due to bycatch in the population or 303 bycatch deaths annually.

As with the observer data, some of the estimates of bycatch mortality from strandings are very high, in the same range as the estimates from observer data. Again the question of their apparent incompatibility with abundance data arises, even if, again, there is substantial uncertainty associated with these estimates. Stranding data could lead to an overestimate of bycatch mortality if bycatch occurred mainly in the coastal zone where carcasses are more likely to wash ashore, although arguably observer data have the opposite bias since they mainly exclude small-scale fishing. It is possible that porpoise abundance has been underestimated due to seasonal movements in and out of the surveyed area, which would probably imply that substantial numbers are present in offshore waters or, perhaps less likely, that there is a substantial amount of movement between Iberia and Africa. Finally, at least until the 2022 (SCANS IV) survey results (Gilles et al. 2023) were published, it seemed plausible that population size had fallen since the 2016 (SCANS III) survey but the abundance estimate from 2022 was around 4000 animals - although, as previously, confidence limits were wide and the estimate was based on a small number of sightings.

The adjacent Northwest African harbour porpoise population in Mauritania presents a low number of strandings due to both the small population size and Mauritanian coastal features that make carcass collection difficult (collection usually involves beach sampling by car), which may produce some biases in the collected stranding data. A coastal monitoring programme carried out along the coast of

Mauritania in 2012 recorded 12 harbour porpoise strandings among other cetacean species strandings. Given the preservation status of the carcasses found, only five could be examined, all of which (100%) showed evidence of bycatch (Mullié et al. 2013). One of three porpoises recorded stranded in Morocco during 2016–2021 by Kaddouri et al. (2023) was bycaught (the cause of death of the other two was unknown) but is unclear whether these would have been Iberian or African porpoises.

Interview surveys

Spain Two interview-based studies in Galicia, by López et al. (2003) and Goetz et al. (2014), generated estimates of total annual cetacean bycatch in Galicia of around 1700 animals. Porpoises were not specifically identified in the earlier study. Goetz et al. (2014) reported that Galician fishers operating fixed gillnets caught an average of 2–3 porpoises per year, with an estimated total annual bycatch by trawl and set gillnet fleets of approximately 40 porpoises, although almost 1300 of the cetaceans bycaught annually were not identified to species. While harbour porpoises were mainly associated with gillnets in Spanish waters, they were also seen close to artisanal gears (those used by the polyvalent fleet), purse seine and beach seine (Goetz et al. 2015).

During 2019 and 2020, the Galician stranding network CEMMA conducted interviews with fishers about their activities in the coastal and adjacent waters from Cape Silleiro to Cape Corrubedo (Galicia) as part of the VIRADA project.[4] The main fishing gears used were trammel nets (*miño*) and single-panel bottom-set gillnet (*beta*), respectively, accounting for 46.7% and 18.1% of all fishing gears used. Of the interviewees, 42.1% reported having had cetaceans, including harbour porpoises, entangled in their nets, mainly in *miños* (Martínez-Cedeira et al. 2021).

Portugal Wise et al. (2007) conducted interviews with skippers during 36 purse seine fishing trips from Portuguese ports to collect information about cetacean sightings and cetacean behaviour around the nets. However, all the reported observations refer to delphinids. Sightings from observer trips carried out in parallel with the interview survey recorded a single porpoise sighting but the only observed bycatch events involved common dolphins.

Face-to-face interviews with skippers were conducted by trained researchers along the whole Portuguese southern coast (Algarve) in the most important landing ports ($n=19$), between March and November 2018, collecting information on incidents of bycatch of marine megafauna (cetaceans, seabirds and seaturtles) in coastal fisheries which took place during 2017. There were reports of common dolphin and bottlenose dolphin bycatch, but none of porpoises being bycaught (Alexandre et al. 2022).

Historical records

Historical records indicate that fishery bycatch mortality has affected small cetaceans, including harbour porpoises, in European waters for centuries (Petitguyot et al. 2024). As noted above, intense conflicts between fisheries and small cetaceans are documented between the eighteenth and twentieth centuries in the Northeast Atlantic and Mediterranean due to animals being blamed for reducing fish catches and for damaging fishing gear when attempting to take fish from the nets (depredation) and when incidentally getting entangled in the nets (bycatch) (Bearzi et al. 2004, Valdés-Hansen 2004, 2009, Mornet 2008, Petitguyot et al. 2024). By damaging nets, both depredation and bycatch events can result in the loss of catch and force the cessation of fishing activities. In earlier times, nets were made of more delicate materials (e.g. cotton and hemp), probably making damage more likely than in the case of modern nylon (plastic) nets. After a bycatch or depredation event, before fishing could resume, nets needed to be repaired on land, and the consequent loss of time and money led to retaliation in the form of culling campaigns (Bearzi et al. 2004, Petitguyot et al. 2024). Tens of thousands of small cetaceans were killed in European waters during this period, but the extent of bycatch-induced mortality of

[4] https://www.programapleamar.es/proyectos/virada-bases-para-la-reduccion-de-la-mortalidad-por-captura-accidental-en-artes-de-pesca

Iberian harbour porpoises is currently unknown. Given that porpoises are apparently frequently caught in artisanal coastal fisheries in the Iberian Peninsula (see below), and similar fishing activities were taking place between the seventeenth and twentieth centuries (Carmona & López Losa 2009, Amorim 2009, López Losa & Amorim 2012), it is likely that Iberian porpoises were often getting entangled.

Spain Small cetaceans were regularly observed at fish markets in Galicia during the nineteenth century, including "toninas" as mentioned by López Ferreiro (1895). In 1991, a porpoise was auctioned at the fish market in Portonovo in Galicia (A. López, Pers. Obs.).

Portugal As described by Teixeira (1979) and Brito and Vieira (2010), harbour porpoises were bycaught in Portugal in the late 1970s and sold in fish markets. A photograph reproduced by Brito and Vieira (2010) shows a bycaught harbour porpoise at the fish market in Caminha (northern Portugal) in 1977 (Figure 14). The animal had drowned in a bottom-set gillnet and (as permitted by the then existing law) was brought to the fish market and auctioned (A. Teixeira, Pers. Comm.)

Sequeira and Ferreira (1994) summarised available information on bycatch mortality of cetaceans in Portuguese fisheries, reporting on landings and effort and mentioning reported bycatches. Killing and sale of marine mammals were legal in Portugal until 1981, when legislation was passed to protect marine mammals, making killing of cetaceans illegal. Fishers then ceased reporting bycatches, and subsequent official records of bycatches were limited. Official records refer to 18 cetaceans (17 common dolphins and one porpoise) killed in trawl nets. Of these, 12 (including the porpoise) were caught in 1980 and the remainder are reported as drowned in nets, five of them during research cruises. Records since 1977 indicated 132 cases of entanglement and 59 deaths in gillnets, although

Figure 14 A bycaught harbour porpoise at the fish market in Caminha (northern Portugal) in 1977, where it was auctioned, as permitted by Portuguese law at the time. (Photograph taken and provided by Antonio Teixeira.)

the authors commented that most such events were not reported. Most of the deaths were thought to be of common dolphins but striped dolphins and porpoises were also regularly taken.

Overview of uncertainty for porpoise bycatch mortality estimates

Pierce et al. (2020, 2022) summarised the available bycatch mortality estimates for the Iberian porpoise population, and these figures have been revised and updated along with newly published information in the present review (Tables 7–10). In order to provide an indication of the precision of the estimates derived from strandings, we estimated the 95% confidence intervals using a combination of non-parametric and parametric bootstrapping, focusing on multi-year estimates with reasonable sample sizes. For the observer estimates we used a non-parametric bootstrap to resample from observer trips in each year, based on 10,000 runs, to estimate the percentage of observed fishing events with bycatch of porpoise and, multiplying by the total number of fishing events, the estimated number of porpoises that were bycaught during all fishing events. The 95% confidence interval was obtained by sorting the 10,000 estimates and extracting the 251st and 9750th values. Values were then summed across years and divided by the number of years to give average values. The averages were then used to derive the annual mortality rate due to bycatch (number bycaught/population size). We also repeated the latter calculation incorporating a parametric bootstrap to simulate abundance estimation, based on the estimate of 2900 animals and associated coefficient of variation from Hammond et al. (2017) and assuming a log-normal distribution (method 2).

For each relevant strandings dataset, we again carried out 10,000 bootstrap runs as follows: (1) resampling with replacement from the sets of carcases examined to obtain the percentage of bycaught animals; (2) resampling with replacement from the set of ages at death (Read et al. 2013, 2020) to estimate the annual mortality rate in the population; (3) using a parametric bootstrap to simulate abundance estimation, based on the estimate of 2900 animals and associated coefficient of variation from Hammond et al. (2017) and assuming a log-normal distribution; (4) estimating the annual population bycatch mortality rate as the product of values from steps *1* and *2* (method 1); (5) estimating the annual number of bycatch mortalities by multiplying the estimate from step *4* by 2900; and (6) estimating the annual number of bycatch mortalities as the product of the estimates from steps *1*, *2* and *3*. For each estimate, again the 95% confidence interval was obtained by sorting the 10,000 estimates and extracting the 251st and 9750th values. These figures are compiled in Table 11, alongside some of the estimates from observer data. Evidently

Table 11 Summary of Bycatch Mortality Estimates (with 95% Confidence Intervals When Available) for Harbour Porpoises in the Iberian Peninsula 1990–2021, Based on (A) Observer Data and (B) Stranding Data

Areas and Years Sampled, Source, (and Subset)	Type of Data	% Bycatch Mortality in Strandings	Estimated Annual Population Bycatch Mortality Rate (%)	Estimated Annual Bycatch Deaths (Method 1)	Estimated Annual Population Bycatch Mortality Rate (Method 2) (%)
(A) Observer Data					
Portugal, 2010–2015 (1)	Observers (purse seines, polyvalent fleet, bottom trawls, long lines, beach seines)	N/A	8.3	241	N/A
Portugal, 2010–2015 (2)	On-board observation of polyvalent, purse seine and trawl fleets	N/A	14.4 (4.6–28.0)	418 (132–813)	14.4 (3.9–37.9)
Portugal, 2010–2020 (2)	On-board observation of polyvalent, purse seine and trawl fleets	N/A	7.9 (2.5–15.3)	228 (47–503)	7.9 (2.1–20.7)

Table 11 (*Continued*) Summary of Bycatch Mortality Estimates (with 95% Confidence Intervals When Available) for Harbour Porpoises in the Iberian Peninsula 1990–2021

Areas and Years Sampled, Source, (and Subset)	Type of Data	% Bycatch Mortality in Strandings (%)	Estimated Annual Population Bycatch Mortality Rate (%)	Estimated Annual Bycatch Deaths (Method 1)	Estimated Annual Bycatch Deaths (Method 2)
(B) Stranding Data					
Galicia, 1990–2010 (3)	Strandings	31 (21.8–41.4)	5.6 (3.73–7.41)	162 (108–215)	162 (73–300)
Galicia, 1990–2019 (4)	Strandings	34.2 (25.8–42.5)	6.2 (4.4–7.9)	180 (128–230	180 (83–324)
Galicia, 1990–2021 (5)	Strandings	40.0 (31.2–48.8)	7.2 (5.3–9.0)	209 (155–262)	209 (100–379)
Galicia, 2000–2020 (6) (PB → NB)	Strandings	41.0 (30.1–51.8)	7.4 (5.2–9.6)	214 (151–276)	214 (99–392)
Galicia, 2000–2020 (6) (PB → *B*)	Strandings	48.2 (37.3–59)	8.7 (6.4–11.0)	252 (185–319)	252 (118–455)
Galicia, 2018–2020 (7)	Strandings	44.0 (22.2–66.7)	8.0 (3.8–12.2)	232 (110–354)	232 (85–469)
Portugal, 2000–2020 (6) (PB → NB)	Strandings	48.8 (42.7–54.4)	8.8 (7.11–10.4)	255 (206–302)	255 (122–452)
Portugal, 2000–2020 (6) (PB → *B*)	Strandings	60.9 (55.2–66.5)	11 (9.1–12.8)	318 (263–372)	318 (155–561)
Portugal, 2018–2020 (7)	Strandings	38.9 (27.8–50.0)	7.0 (4.8–9.3)	203 (139–269)	203 (92–373)
Galicia + N. Portugal, 1990–2010 (3)	Strandings	44.7 (37.1–52.2)	8.0 (6.2–9.8)	233 (180–283)	233 (110–413)
Galicia + Portugal, 2000–2020 (6) (PB → NB)	Strandings	47.0 (41.8–52.2)	8.5 (6.9–9.9)	245 (200–287)	245 (119–432)
Galicia + Portugal, 2000–2020 (6) (PB → *B*)	Strandings	58.0 (52.7 62.9)	10.4 (8.6–12.1)	302 (251–351)	302 (148–529)

The bootstrap methodology used to derive 95% confidence intervals is described in the text. For data from Read et al. (2013, 2020) and Pierce et al. (2020), known bycatches handed in by fishers were removed from the dataset prior to calculations of bycatch rate. Torres-Pereira et al. (2023a) distinguished (certain) bycatches and probable bycatches. The two sets of estimates refer to when probable bycatch (PB) was treated as (1) non-bycatch (NB) and (2) bycatch (B), respectively. "Method 2" for calculating 95% confidence intervals includes consideration of uncertainty in the population size estimate.

Sources: (1) Vingada and Eira (2018) (MARPRO project), (2) ICES (2013a,b, 2014c, 2017), (3) Read et al. (2013, 2020), Pierce et al. (2020), (5) CEMMA, unpublished data, (6) Torres-Pereira et al. (2023a), (7) ICES (2020c, 2021c), (8) Goetz et al. (2014), (9) Martinez-Cedeira and Lopez (2018), (10) Vingada et al. (2011).

this process does not speak to the accuracy of the estimates, and biases could arise from several sources, including unrepresentative strandings data and changing population size, age structure and mortality rate over time.

As seen in the previous tables and text, the majority of estimates suggest a mortality of a few hundred individuals per year due to bycatch although observer data during 2016–2020 included no records of porpoise bycatch. Strandings continue to include a substantial proportion of bycaught animals, suggesting that on-board observation is not adequately documenting porpoise bycatch, possibly because it mainly occurs as a result of small-scale fisheries. It should be noted that the various strandings datasets all draw on data from the same strandings networks and are thus not all independent estimates. Furthermore, although the estimates from strandings based on data presented by Read et al. (2013, 2020) and Pierce et al. (2020) excluded known bycatches (animals handed in by fishers) from the "strandings" datasets – and there were in any case few such animals – it is not clear whether such animals had been excluded from the other compilations of strandings data. If not, in these cases there would be a small upward bias in the estimated proportion of bycatches among stranded animals.

The 95% confidence limits on the annual bycatch mortality rate and annual number of bycatch deaths are wide (very wide indeed for the observer data, as expected given very low observer coverage) but even the lower 95% confidence limits for the annual mortality rate due to bycatch are always higher and often much higher than the removal threshold of 1.7% mortality due to anthropogenic causes proposed by ASCOBANS (2000) (i.e. 49 animals from a population of 2900), thus also exceeding the annual PBR-based limit of 25 animals calculated during the 2018 IMR/NAMMCO workshop and obviously exceeding the bycatch limit of zero proposed by OSPAR.

Hunting

Harbour porpoises are currently protected by law in both Spain and Portugal, as well as under EU directives, and as such no exploitation is permitted. However, this is a relatively recent development. There is a long history of human exploitation of small cetaceans worldwide, and the Iberian Peninsula is no exception.

Although zooarchaeological remains of harbour porpoise are known from elsewhere in Europe (Evans & Mulville 2018), to our knowledge, there are no published records from the Iberian Peninsula. Porpoise remains dating from the Mediaeval period were recovered in England and the Netherlands (van den Hurk et al. 2020, 2021). In France, remains dating from pre-Roman times (Oueslati 2017, in Bernal-Casasola 2018) and historical records from Mediaeval to modern times attest to long-lasting porpoise exploitation (Berthelot 1840, Musset 1964, Fichou & Levasseur 2004). In the Mediaeval period, cetaceans were "fished" to eat on those days when the church did not permit eating of meat. In England, porpoise with frumenty (a porridge) was served at the wedding feast of Henry IV in 1404; roast porpoise was served at the crowning of Henry V; and porpoise with a sauce made of bread crumbs, vinegar and sugar was one of the favourite foods of Henry XIII, which could be eaten on fast-days because it was considered to be a fish (Fosså 1995). Dillen (2022) describes the annual supply of a porpoise (meerzwijn) for the consumption of the aldermen of Bruges in late Mediaeval times. Records indicate that porpoises were intensively hunted from at least the fourteenth century to the twentieth century in Denmark (Kinze 1995) and that hunting took place in other places in the Northeast Atlantic and Arctic (Stenson 2003) as well as in the Black Sea (Tonay & Öztürk 2012). The sale of a porpoise, alongside two seals and various fish and shellfish, is illustrated in Flemish painter Frans Snyders' painting "The Fish Market" from 1618, while his "Fish Stall" from the same period includes a porpoise, a seal and an otter (Figure 15).

In the Iberian Peninsula, remains of dolphins have been found dating from the Middle and Upper Paleolithic, and remains of other, larger, cetacean species have been found dating from the Late Neolithic/Chalcolithic and Roman periods (Corchón-Rodríguez & Álvarez-Fernández 2008, Mariezkurrena-Gastearena 2011, Álvarez-Fernández 2015, Moreno-García et al. 2017, Benito et al. 2019). Given the rarity of identifiable cetacean remains in the zooarchaeological record (see Speller

Figure 15 Photograph of the oil painting *Fish stall* by Frans Snyders and Jan Wilden, illustrating a stall in the fish market of Antwerp, dated between 1618 and 1621, held at the State Hermitage Museum, St Petersburg, Russia. Among the fish and shellfish for sale are several tortoises, a porpoise, a seal and an otter. Available at: https://commons.wikimedia.org/wiki/File:Frans_Snyders_-_Fish_Stall_-_WGA21521.jpg

et al. 2016, Evans & Mulville 2018), one cannot exclude the possibility of harbour porpoises being exploited in the area during prehistoric and historical times. The coastal distribution of the species would have made it easy for humans to exploit at least beached individuals, and/or those approaching the shoreline. More recent historical information testifies to the exploitation of the species in both Spain and Portugal.

Mediaeval Portuguese historical records from as early as the thirteenth century suggest that *toninhas* were captured and consumed in large numbers for centuries, mainly during the late nineteenth century and much of the twentieth century (Brito & Veira 2010). Although many of these records probably refer to common dolphins, some of the historical accounts cited in Brito and Veira (2010) and Brito (2011) distinguished between *golfinho* (dolphin) and *toninha*, which suggests that porpoises were also exploited. There is also historical evidence of capture of small cetaceans, including harbour porpoises, for human consumption in coastal waters of Northern Spain from as early as the Middle Ages (Valdés Hansen 2004). Records collated by Valdés Hansen (2004, 2009) refer to intense conflicts between fisheries and small cetaceans occurring from Galicia to Cantabria between the seventeenth century and the first half of the twentieth century. Small cetaceans were considered to be direct competitors that depleted fish stocks, damaged the nets by getting themselves entangled and ate the entangled fish. These conflicts resulted in the killing of many cetaceans, including harbour porpoises (Valdés Hansen 2004).

Surveys undertaken along the Portuguese coast between 1976 and 1978 revealed that small cetaceans were captured opportunistically using hand harpoons (and sometimes taken as bycatch) and sold in beach fish markets, as part of a local non-industrial fishery (Teixeira 1979, Brito & Veiera 2010). Most captures were of common dolphin but a harbour porpoise was caught and sold at Cascais market in 1979 (Teixeira 1979, in Brito & Veira 2010). Sequeira (1996) noted that, until

1981, it was legal for porpoises to be caught and sold in Portugal but she considered that they were captured incidentally rather than deliberately targeted.

Examination of carcasses of harbour porpoises stranded in Galicia over the last 30 years has not revealed any evidence of hunting or of deliberate killing by fishers (e.g. perforations in the carcasses) or evidence that flesh was removed for human consumption (e.g. dorsal muscles removed). However, López et al. (2003) reported that 69 fishers interviewed in Galicia in the late 1990s referred to cetaceans being used for human consumption, although the species involved were not identified. Some interviewees said they had eaten cetaceans (fillets or the liver) and others commented that cetaceans were eaten in the Basque country, Portugal and France. The use of cetaceans for bait, animal food and as a source of fat was also mentioned. It is not clear whether these animals were bycatches or were deliberately killed, or whether they included porpoises. In France, it is documented from interviews with retired fishers that they would catch a dolphin or a porpoise after 15–20 days at sea, when meat supplies had run out (Anonymous, Pers. Comm.).

Informal interviews were conducted (by M. Petitguyot) with six retired fishers in August 2022 in the town of A Guarda in the south of Galicia to enquire about consumption of harbour porpoises in the past. The fishers used small boats (*gamelas*) to target various types of fish (e.g. monkfish (*rape*), turbot (*rodaballo*) and shellfish). Although they never hunted porpoises, they would consume porpoises that were bycaught in their gillnets (*trasmallos* and *miños*). Bycaught porpoises represented an easily accessed source of protein that was particularly welcome in periods when food availability was sometimes limited (i.e., during the Spanish Civil War, 1936–1939). The meat was directly consumed by the family or close friends, but not sold to others or in markets. Although one interviewee had eaten porpoises only twice in his life, the others had consumed porpoise regularly when they were actively fishing (one mentioned eating around five porpoises per year). The fishers mentioned that Portuguese fishers also ate porpoises. The loin (*lomo*) was the most commonly eaten part of the animal; the lomo was soaked in water for a few days in order to remove blood and an excessive taste of salt. Once cleaned, it was cooked in various dishes or transformed into cured meat. Boiled lomo was used to make stews, cooked with fish oil (e.g. from sardines) and accompanied by potatoes and peppers. Lomo cured in salt (*salazon*) would be cut into thin slices (similar to *jámon ibérico*). The ribs were left in salt for a whole month to remove the blood and then used to make *cocido* (a stew), eaten with grains or legumes (e.g. chickpeas, haricots, beans, lentils and rice) and other vegetables. The liver was cut into fillets and fried with garlic and salt.

Pollutants and other harmful substances

Cetaceans are exposed to a range of harmful substances of anthropogenic and natural origins. Among the POPs, historically pesticides such as dichlorodiphenyltrichloroethane (DDT) were a major concern but more recently PCBs have attracted most attention, not least due to their known effects on the immune system and on reproduction in mammals. Other potentially harmful organic chemicals include polyaromatic hydrocarbons (PAHs), which are associated with oil pollution, while heavy metals are among the most relevant inorganic chemicals. Aguilar and Borrell (1995) reviewed published information on organic contaminants and metals in harbour porpoise in the eastern North Atlantic (from France to the Baltic Sea, 1967–1991) and noted that levels of organochlorines, especially PCBs, were high enough to cause concern about their effect on population status.

POPs

PCB-induced infertility was demonstrated experimentally in seals (Reijnders 1986). Based on results of studies on mink, otters and seals, a total PCB concentration of 17 μg g^{-1} (i.e. 17 mg kg^{-1}) lipid weight in blubber was estimated to be the threshold level for the onset of physiological effects (including those on reproduction) in aquatic mammals (Kannan et al. 2000), and this value has been applied in various studies on cetaceans (e.g. Schwacke et al. 2002, Pierce et al. 2008, Méndez-Fernandez et al. 2014a). This threshold was based on the commercial PCB mixture Arclor 1254 and was considered to be equivalent to 9 μg g^{-1} lipid weight in blubber when considering the sum of 25 CBs (namely, CBs 18, 28, 31, 44, 47, 49, 52, 66, 101, 105, 110, 118, 128, 138, 141, 149, 151, 153, 156, 158, 170, 180, 183, 187 and 194) (Murphy et al. 2015, Jepson et al. 2016). Jepson et al. (2016) commented that mink are especially sensitive to PCBs, and this threshold may be too low for cetaceans. Based on the analysis of concentrations of 25 PCBs in female harbour porpoises, Murphy et al. (2015) proposed that ΣPCB concentrations over 11 μg g^{-1} in mature females should be considered as indicating infertility or reproductive failure. Jepson et al. (2016) proposed the use of 41 μg g^{-1} as the threshold for reproductive impairment in cetaceans, based on the results for ringed seals in which this concentration was associated with profound reproductive impairment (Helle et al. 1976). In their study of PCB levels in porpoises, Van den Heuvel-Greve et al. (2021) argued that the most relevant congeners are 47, 49, 52, 101, 105, 118, 128, 138, 149, 151, 153, 156, 170, 180, 187, 194 and 202, thus deriving a sum-17PCB, also noting the value of calculating a sum-7PCB for comparison with OSPAR and ICES data, based on CBs 28, 52, 101, 118, 138, 153 and 180 (OSPAR Commission 2016). There has thus been some inconsistency in relation to which CBs have been measured and total concentrations should be treated as thresholds for the onset of reproductive impairment and for complete infertility, and this may of course differ according to species and indeed in relation to other factors such as health status. In relation to effects of POPs on cetacean reproduction, the focus has been almost exclusively on females. However, it has been shown that PCBs also cause reduced testis weight in male harbour porpoises and hence, presumably, also reduces male fertility (Williams et al. 2021).

Results on average PCB concentrations recorded in harbour porpoises in the North Atlantic are compiled in Table 12A. Comparisons are difficult given differences in the number and identities of congeners included in total PCB concentrations (see Table 12B), as well as expected differences related to age, sex and reproductive status of the studied animals. Where possible we included the total concentration of the ICES 7 congeners, thus permitting comparison with the Kannan et al. (2000) threshold of 17 μg g^{-1} lipid, which is equivalent to Σ [ICES 7 CBs] = 5.67 μg g^{-1} lipid (Jepson et al. 2005, Pierce et al. 2008, Murphy et al. 2010). Since the 17 μg g^{-1} threshold based on Aroclor 1254 was considered to be equivalent to 9 μg g^{-1} of the 25 CBs measured by Jepson et al. (2016), it follows that the other thresholds based on 25 CBs suggested by Murphy et al. (2015) and Jepson et al. (2016), i.e., 11 μg g^{-1} and 41 μg g^{-1}, would be equivalent to 6.93 μg g^{-1} and 25.81 μg g^{-1}, respectively, of the ICES 7 CBs.

The EU-funded BIOCET project (2001–2004) surveyed pollutant levels in tissues of small cetaceans on European Atlantic coasts. The animals sampled from Galicia included three porpoises. Total PCB burden was estimated as 3 × Σ [ICES 7 CBs]. One of these individuals had a total PCB concentration in blubber higher than the threshold of 17 μg g^{-1} lipid weight. Of the five cetacean species from the NWIP studied by Méndez-Fernandez et al. (2014a), bottlenose dolphin and harbour porpoise showed the highest concentrations of PCBs in their blubber. In these species, 100% and 75%, respectively, of the individuals analysed exceeded the toxic threshold of 17 μg g^{-1} lipid weight.

Table 12A Mean PCB Concentrations (± Standard Deviation and/or with Range (in Parentheses) When Given) Reported for Harbour Porpoises in the North Atlantic

Area	Time Period	Sex/Maturity	Sum [PCB] (μg.g⁻¹ Lipid Weight)	ICES 7 [PCB] (μg.g⁻¹ Lipid Weight)	No of Porpoises	No of Congeners	Reference
Canada (Bay of Fundy / Gulf of Maine)	1989–1991	Males	17.28 ± 11.18		55	68	Westgate et al. (1997)
		Females	11.38 ± 4.81		53	68	
Canada (Gulf of St. Lawrence)		Males	10.64 ± 5.43		31	68	
		Females	7.15 ± 3.85		31	68	
Canada (Southeast Newfoundland)		Males	5.24 ± 2.51		18	68	
		Females	5.49 ± 4.37		11	68	
Denmark and Norway	1987–1991	Males	23.27 (3.75–65.26)		34	47	Klevaine et al. (1995)
UK	1990–2012	Males	19.41 (0.44–150.47)	10.14 ± 10.42 (0.25–65.88)	146	23	Jepson et al. (2016)
		Females	13.49 (0.40–159.68)	6.89 ± 8.16 (0.18–71.1)	134	23	
	2000–2003	Females	10.52 ± 13.15	6.77 ± 8.41 31% above threshold	31	16	Pierce et al. (2008)
	2014–2018	Males and Females	16.31 (0.46–159.68)		604	25	Williams (2021)
Ireland	2000–2003	Females	5.35 ± 4.75	3.51 ± 3.15 25% above threshold	12	16	Pierce et al. (2008)
Netherlands and Belgium	2000–2003	Females	15.02 ± 8.57	10.20 ± 6.00 74% above threshold	19	16	Pierce et al. (2008)
Netherlands	1999–2004	Males and Females	12.4		35	21	Weijs et al. (2009)
	2008–2019	Both sexes	12.1 ± 16.42 (0.2–90.2) 38.5% above threshold	8 ± 10.64 (0–55.4)	121	17	Van den Heuvel-Greve et al. (2021)
		Foetuses	3.46 ± 3.10 (0.6–8.8) 0% > above threshold	2.39 ± 2.24 (0–6.3)	9	17	

(Continued)

Table 12A (*Continued*) Mean PCB Concentrations (± Standard Deviation and/or with Range (in Parentheses) When Given) Reported for Harbour Porpoises in the North Atlantic

Area	Time Period	Sex/Maturity	Sum [PCB] (µg.g^{-1} Lipid Weight)		ICES 7 [PCB] (µg.g^{-1} Lipid Weight)	No of Porpoises	No of Congeners	Reference
		Neonates	11.50±7.10 (1.8–24.0)		8.19±5.19 (1.4–17.1)	16	17	
		Juveniles	15.30±20.22 (2.6–90.2) 48.9% above threshold		9.8±12.67 (1.7–55.4)	45	17	
		Mature Males	23.50±15.46 (3.8–62.4) 92.3% above threshold		16.24±10.5 (2.6–42.2)	13	17	
		Mature Females	6.70±13.47 (0.2–76.2) 10.5% above threshold		4.45±8.94 (0.1–49.8)	38	17	
Belgium and France (Southern North Sea)	2010–201	Immature Female	32±21			3	20	Mahfouz et al. (2014)
		Immature Male	20±31			12	20	
		Mature Females	4±1.8			4	20	
		Mature Males	22			1	20	
France	2000–2003	Females		13.81±10.58	9.20±6.96 50% above threshold	2	16	Pierce et al. (2008)
Spain (Northwest)	2000–2003	Females		5.31±4.20	3.42±2.66 33% above threshold	3	16	Pierce et al. (2008)
	2004–2008	Immature Males		9.4±3		3	32	Méndez-Fernandez et al. (2014a)
		Immature Females		10.8±2.8		5	32	
		Mature Males		50.8		1	32	

(Continued)

Table 12A (*Continued*) Mean PCB Concentrations (± Standard Deviation and/or with Range (in Parentheses) When Given) Reported for Harbour Porpoises in the North Atlantic

Area	Time Period	Sex/Maturity	Sum [PCB] (μg.g^{-1} Lipid Weight)	ICES 7 [PCB] (μg.g^{-1} Lipid Weight)	No of Porpoises	No of Congeners	Reference
		Mature Females	37.5±30.8		3	32	
	2015–2019	Immature Males	6.9±6.7	5.1±5.2	4	14	TRANSITION project, unpublished data
		Immature Females	9.1±6.2	6.8±4.7	4	14	
		Mature Males	12.62±9.8	9.2±6.9	4	14	
		Mature Females	14.2±10.6	10.9±8.5	2	14	

We give results in μg.g^{-1} (equivalent to mg.kg^{-1}). Also indicated are the number, sex and maturity stage of the individuals analysed, number of PCB congeners analysed, and the source of the information. The sum [PCB] columns refers to the set of congeners measures in each study (see Table 8B). To facilitate comparison, where possible we include the summed concentration of the ICES 7 PCBs. Previously reported thresholds correspond to Σ[ICES7 PCBs] of 5.67 μg.g^{-1} (Kannan et al. 2000), 6.93 mg kg^{-1} (Murphy et al. 2015) and 25.81 μg.g^{-1} (Jepson et al. 2016). Pierce et al. (2008) and Van den Heuvel-Greve et al. (2021) reported on the percentage of animals with PCB concentrations above the Kannan et al. (2000) threshold. Means and standard deviations of total PCB concentration in the latter study were reconstructed from their supplementary data

Table 12B PCB Congeners Measured in the Different Studies Included in Table 12A

Congener	ICES 7	Klevaine et al. (1995)	Westgate et al. (1997)	Pierce et al. (2008)	Weijs et al. (2009)	Méndez-Fernandez et al. (2014a)	Jepson et al. (2016)	Williams (2021)	van den Heuvel-Greve et al. (2021)	Transition project (unpubl.)
18							1	1		
26		1								
28	1			1	1	1	1	1		1
31					1	1	1	1		1
44						1	1	1		
47		1					1	1	1	
49				1		1	1	1	1	
52	1	1	1	1	1	1	1	1	1	1
66		1					1	1		
70		1				1				
74					1	1				
87		1								
91		1								
95		1	1		1					
97						1				
99		1			1	1				
101	1	1	1	1	1	1	1	1	1	1
105					1	1	1	1	1	1
107		1								
110		1			1	1	1	1		
111		1								

(Continued)

Table 12B (*Continued*) PCB Congeners Measured in the Different Studies Included in Table 12A

Congener	ICES 7	Klevaine et al. (1995)	Westgate et al. (1997)	Pierce et al. (2008)	Weijs et al. (2009)	Méndez-Fernandez et al. (2014a)	Jepson et al. (2016)	Williams (2021)	van den Heuvel-Greve et al. (2021)	Transition project (unpubl.)
114		1				1				
118	1	1	1	1	1	1	1	1	1	1
123		1				1				
128		1		1	1	1	1	1	1	
130		1								
132		1				1				
137		1				1				
138	1	1	1	1	1	1	1	1	1	1
141		1		1			1	1		
146		1								
148		1								
149			1	1	1	1	1	1	1	1
151		1	1	1			1	1	1	
153	1	1	1	1	1	1	1	1	1	1
156					1	1	1	1	1	1
157		1				1				
158		1				1	1	1		
167		1				1				
170		1		1	1	1	1	1	1	1
174		1								
177		1								

(Continued)

Table 12B (*Continued*) PCB Congeners Measured in the Different Studies Included in Table 12A

Congener	ICES 7	Klevaine et al. (1995)	Westgate et al. (1997)	Pierce et al. (2008)	Weijs et al. (2009)	Méndez-Fernandez et al. (2014a)	Jepson et al. (2016)	Williams (2021)	van den Heuvel-Greve et al. (2021)	Transition project (unpubl.)
178		1								
179		1								
180	1	1	1	1	1	1	1	1	1	1
183		1		1	1	1	1	1		
185		1								
187		1	1	1	1	1	1	1	1	1
189						1				
192		1								
193		1								
194		1		1	1	1	1	1	1	1
195		1								
196		1								
199					1					
201		1								
202		1							1	
206		1								
209		1				1				
SUM	7	47	68	16	21	32	25	25	17	14

The number 1 indicates that a congener was measured. Note that most studies report the ICES 7 congeners (shaded cells). Westgate et al. (1997) analysed 68 congeners but only mentioned those that together comprised around 50% of the total. Pierce et al. (2008) measured concentrations of 18 congeners but the study included UK porpoise samples for which only 16 congeners had been measured (the two missing CBs were 99 and 177) and, for comparability, results for 16 congeners in porpoises were presented for all regions.

Recent unpublished work conducted in part under the TRANSITION project (Gutiérrez-Muñoz et al. in prep) found somewhat higher total PCB concentrations in porpoises stranded in Galicia during the period 2015–2023 ($N=19$) than found by Méndez-Fernandez et al. (2014a) during the period 2004–2008. Although further work is needed as sample sizes were small, a decreasing trends over time has been reported elsewhere in Europe (e.g. Williams et al. 2023).

Numerous other Persistent Organic Pollutants are found in cetaceans, including insecticides (e.g. DDTs) brominated flame retardants. The three Galician porpoises included in the Pierce et al. (2008) study had low concentrations of flame retardants in their blubber, namely, polybrominated diphenylethers (PBDEs) (Pierce et al. 2008) and hexabromocyclododecane (HBCD) (Zegers et al. 2005). HBCD concentrations were lower than those in porpoises sampled from Dutch, Irish and Scottish waters. While adverse effects of pesticides on marine animals are well known, the effects of some other POPs remain undescribed. We are not aware of studies on other organic pollutants in Iberian porpoise but it may be noted that in Norwegian and Danish waters, organochlorine pesticide concentrations in porpoises were two to three times higher than those in seals in samples collected during 1987–1991 (Klevaine et al. 1995).

Other contaminants

The proximity of Galicia to one of the world's busiest shipping routes, along with the presence of a refinery located in the port of A Coruña, makes the NWIP coastline particularly vulnerable to oil pollution. Galicia has experienced five out of the eleven major oil spills in Europe in the last three decades (Franco et al. 2006, González et al. 2006, Loureiro et al. 2006). The "Prestige" oil spill in November 2002 released 60,000 metric tonnes of oil into the Atlantic off Galicia and polluted 1300 km of coastline (Loureiro et al. 2006). In the 6 months following the spill, 124 cetaceans were stranded along the Galician coast, of which 35% were oiled and 3% were considered to have died as a direct result of oil. This included the mortality of at least one porpoise (López et al. 2005, Loureiro et al. 2006). It is also likely that there were indirect impacts on porpoises via the food chain. Sánchez et al. (2006) monitored the abundance of four key benthic and demersal species and noted reductions in the abundance of three of them: Pandalid shrimp (*Plesionika heterocarpus*), Norway lobster (*Nephrops norvegicus*) and four-spot megrim (*Lepidorhombus boscii*), although the abundance and distribution of juvenile hake, which are eaten by porpoises, showed no significant changes.

Coastal upwelling systems often experience the occurrence of harmful algal blooms (HABs), e.g., in cultivated mussels (Álvarez-Salgado et al. 2008). The main associated risk to cetaceans relates to the production of natural toxins by various species of phytoplankton and the accumulation of these toxins in food webs (Broadwater et al. 2018, Fire et al. 2021). Domoic acid was found in harbour porpoises (and in a porpoise foetus) from Alaska by Lefebvre et al. (2016), while saxitoxins were reported in porpoises from Canada by Bates et al. (2020). Another toxin associated with HABs is the neurotoxin β-N-methylamino-L-alanine (BMAA), which is produced by both cyanobacteria and diatoms. Soliño et al. (2022) found no evidence of BMAA in stranded porpoises, bottlenose dolphins and common dolphins from Northwest Spain.

Concerning inorganic contaminants, Lahaye et al. (2007) reported low concentrations of the toxic chemical elements mercury (Hg) and cadmium (Cd) in porpoises from Galicia. According to Méndez-Fernandez et al. (2014b), the concentrations of Hg and Cd found in Iberian toothed whales indicate that these populations are not especially threatened by Hg and Cd exposure in the area. Compared with the values reported by Méndez-Fernandez et al. (2014b), 42 harbour porpoises stranded in Portugal from 2005 to 2013 had higher levels of Hg in liver and lower levels of Cd in kidneys (Ferreira et al. 2016). The higher Hg levels may reflect anthropogenic sources, with Portuguese animals inhabiting waters closer to the Mediterranean where high levels of Hg occur in the seawater due to human activities, such as mercury mining, and a high density of sub-marine volcanoes (Bernhard & Renzoni 1977, Cossa et al. 1997, 2022). Nevertheless, the recorded hepatic Hg levels did not exceed the level for toxic thresholds in marine mammals of 60 μg g^{-1} wet weight (Rawson et al. 1993).

The health implications of the presence of microplastics in cetacean stomachs are presently unknown. Recent studies on plastics in Portugal have shown that the environmental concentration of microplastics is higher in the northern and central coasts of Portugal, although these concentrations are low to moderate compared with other areas of the NE Atlantic (Prata et al. 2020). In the same study, microplastics were found in most of the species of the biota groups investigated (loggerhead turtles, mussels, commercial fish and birds), although with differences in prevalence (Prata et al. 2020). Guilhermino et al. (2021) analysed several species of fish from the estuary of the Minho, on the northern border of Portugal with Spain. Ninety-four percent of the fishes sampled presented microplastics, with a mean of 8 ± 7 microplastic items/fish (among the highest described in the literature at worldwide level). A study of microplastics in anchovy (*Engraulis encrasicolus*), sardine, common dragonet and the mullet *Mullus surmuletus* in Galician waters also showed a high incidence of microplastics (78%) in the fish examined (Filgueiras et al. 2020). Two of these species have been recorded in porpoise diet in Galicia (see section 'Feeding ecology – Diet and prey consumption'). There have been no studies on microplastics in the stomach contents of porpoises in the Iberian Peninsula but a recent study on common dolphin stomach contents in Galicia found microplastics in every stomach ($N = 35$) examined (Hernández-González et al. 2018). It would be surprising if this was not also the case for porpoises.

Currently, there is no routine monitoring of contaminants in stranded cetaceans in the Iberian Peninsula. The studies reported above have been based on short-term, usually project-based funding. Consequently, the absence of reports does not necessarily imply the absence of a problem.

Pathogens: viruses, bacteria and parasites

Viruses

Several viruses have been detected in harbour porpoises in Northeast Atlantic waters, including rhabdovirus, papillomavirus, herpesvirus and morbillivirus (Van Bressem et al. 1999, 2014). Both herpesvirus and morbillivirus can cause serious and lethal disease. Cetacean morbillivirus infections result in a high mortality rate and, due to recurrent epizootics, can have long-term effects on population dynamics in cetaceans (Van Bressem et al. 1999). Morbillivirus infections in cetaceans were first described at the time of the 1988 phocine distemper virus (PDV-1) epidemic among harbour seals, when morbillivirus antigens were detected in organs of harbour porpoises that had died on the Irish coast (Kennedy et al. 1988, McCullough et al. 1991), although exposure in cetaceans may have occurred prior to the PDV outbreak (Härkönen et al. 2006). A morbillivirus epidemic was subsequently seen in striped dolphins (*Stenella coeruleoalba*) in the western Mediterranean, from which a dolphin morbillivirus (DMV) was isolated from striped dolphins (Van Bressem et al. 1991). Visser et al. (1993) isolated a (previously unidentified) porpoise morbillivirus (PMV) from two harbour porpoises that died in the Dutch Wadden Sea (North Sea) in 1990. Porpoise and dolphin morbilliviruses represent a distinct lineage closer to ruminant morbilliviruses than to carnivore morbilliviruses (Barrett et al. 1993).

Herpesvirus infections have previously been described in harbour porpoises from Sweden and the Netherlands. Kennedy et al. (1992) reported herpes viral encephalitis in a porpoise stranded in Sweden, while van Elk et al. (2016) examined stranded porpoises in the Netherlands, detecting three types of herpesvirus. A known gammaherpesvirus (PPHV-1) was found in one animal (out of 117 animals), associated with a genital plaque, and two novel alphaherpesviruses. PPHV-2 was found in the brain tissue of one animal (out of 74) associated with encephalitis, while PPHV-3 was found in the brain (4 of 74 animals), blowhole swabs (2 of 43 animals) and genital swabs (2 of 43 animals) but not associated with disease. The authors note that PPHV-2 resemble dolphin herpesviruses, while PPHV-3 was more similar to cervid herpesviruses.

In Portugal, samples from two out of 31 stranded porpoises tested positive for different alphaherpesviruses (representing two out of three monophyletic branches detected among the cetacean samples). In addition, in a phylogenetic analysis, a third porpoise sample clustered with gammaherpesvirus sequences (Bento et al. 2019).

Bacteria

Bacterial species detected in harbour porpoises stranded along the Northeast Atlantic coast include *Actinobacillus delphinicola*, *Actinobacillus scotiae*, *Brucella* spp., *Candida albicanis*, *Cetobacterium ceti*, *Nocardia asteroides*, *Salmonella* and *Streptococcus* spp. (Higgins 2000). *Brucella ceti* appears to be present in cetaceans globally, and distinct dolphin and porpoise types have been identified. Brucellosis in dolphins and porpoises may result in significant clinical and pathological signs related to abortions, male infertility, neuropathogenicity, cardiopathies, bone and skin lesions and death (Guzmán-Verri et al. 2012). Dagleish et al. (2008) and Jauniaux et al. (2010) described pathologies associated with *Brucella* infection in a male porpoise and a female porpoise, respectively, and their findings suggest that infection may result in sterility. Murphy et al. (2015) noted that a porpoise from the UK with a *Brucella ceti* of the mammary glands had lost its calf, possibly due to abortion. *Brucella* was isolated from a striped dolphin live-stranded on the coast of Cantabria (northern Atlantic coast of Spain) in 2004 (Muñoz et al. 2006). Regional variations in pathologies related to bacterial flora were described by Siebert et al. (2009), showing less bacterial growth associated with fewer pathologies in harbour porpoises from Greenlandic, Icelandic and Norwegian waters, compared to individuals from German North and Baltic Seas.

To the authors' knowledge, the only study reporting on bacteria in porpoises stranded along the Iberian coastline was conducted by Soares-Castro et al. (2019) who carried out a comparative analysis of the oral microbiome of the three most common cetacean species stranded on the Northwestern Atlantic Iberian coast (common dolphin, striped dolphin and harbour porpoise). Microbiomes are expected to include both beneficial and potentially harmful microbes. This study revealed significant differences in microbiota between the Phocoenidae and the two Delphinidae species. The Shannon diversity index for harbour porpoise microbiomes was lower than that in common dolphins and higher than that in striped dolphins. The high heterogeneity between species was reflected in the small number (12) of Operational Taxonomic Units (OTU) present in all samples from all three species. Of the 903 OTUs described in the porpoise samples, 18 were present in all porpoise samples (see Table 13). Two bacterial genera were suggested as "fingerprints" for porpoises (*Dethiosulfovibrio* and *Marinicella*), which could be used as bioindicators to develop diagnostic assays and monitoring tools for the assessment of cetacean population and ecosystem health. The microbial community also differed according to age class, with adult and mature animals showing a higher mean number of OTUs and a higher Shannon index. Differences between males and females were detected only at family level. Microbiomes differed depending on the location and health status of the animals, although porpoise samples were insufficiently variable to highlight patterns in this species.

Parasites

Cetaceans host a wide range of parasites. The most visible endoparasites during necropsy include nematodes in the respiratory and gastrointestinal tracts, which are often present in large numbers. Odontocete cetaceans are final hosts of all nine known species of the gastrointestinal parasite *Anisakis* (Mattiucci et al. 2018). Adult *Anisakis* usually live unattached within the forestomach (Smith 1989, Herreras et al. 2004) and feed off the food bolus, something that has been associated with anaemia in heavily infected hosts (Gibson et al. 1998). The prevalence of *Anisakis* spp. is apparently increasing in cetaceans in some areas of the Northeast Atlantic (Lino et al. 2022), where the high abundance of *Anisakis* larvae found in some commercial fish species such as European

Table 13 Taxonomy of the Operational Taxonomic Units (OTUs) Composing the Core Microbiome of the Oral Cavity of the Sampled Harbour Porpoises

Phylum	Class	Order	Family	Genus	Species
Proteobacteria	Gammaproteobacteria	Pasteurellales	Pasteurellaceae	*Phocoenobacter*	Uncultured bacterium
Fusobacteria	Fusobacteriia	Fusobacteriales	Fusobacteriaceae	*Fusobacterium*	Uncultured bacterium
Fusobacteria	Fusobacteriia	Fusobacteriales	Leptotrichiaceae	*Oceanivirga*	Uncultured bacterium
Bacteroidetes	Bacteroidia	Bacteroidales	Porphyromonadaceae	*Porphyromonas*	Uncultured bacterium
Proteobacteria	Gammaproteobacteria	Cardiobacteriales	Cardiobacteriaceae	uncultured	Uncultured bacterium
Proteobacteria	Epsilonproteobacteria	Campylobacterales	Campylobacteraceae	*Arcobacter*	Uncultured bacterium
Firmicutes	Clostridia	Clostridiales	Lachnospiraceae	uncultured	Uncultured bacterium
Bacteroidetes	Flavobacteriia	Flavobacteriales	Flavobacteriaceae	*Maritimimonas*	Uncultured bacterium
Proteobacteria	Gammaproteobacteria	Pasteurellales	Pasteurellaceae	*Actinobacillus*	Uncultured bacterium
Proteobacteria	Gammaproteobacteria	Vibrionales	Vibrionaceae	*Vibrio*	*Vibrio* sp. S-C1-5
Proteobacteria	Epsilonproteobacteria	Campylobacterales	Campylobacteraceae	*Campylobacter*	Uncultured bacterium
Firmicutes	Clostridia	Clostridiales	Family XIII	uncultured	Uncultured bacterium
Proteobacteria	Betaproteobacteria	Burkholderiales	Burkholderiaceae	*Ralstonia*	Uncultured bacterium
Firmicutes	Clostridia	Clostridiales	Family XII	*Fusibacter*	Uncultured bacterium
Proteobacteria	Gammaproteobacteria	Cardiobacteriales	Cardiobacteriaceae	uncultured	Uncultured bacterium
Bacteroidetes	Bacteroidia	Bacteroidales	Bacteroidales S24-7 group	uncultured bacterium	Uncultured bacterium
Bacteroidetes	Bacteroidia	Bacteroidales	Bacteroidales S24-7 group	uncultured bacterium	Uncultured bacterium
Fusobacteria	Fusobacteriia	Fusobacteriales	Fusobacteriaceae	*Fusobacterium*	Uncultured bacterium

Source: From Soares-Castro et al. (2019).

hake and blue whiting is a cause for concern for human consumers (see Levsen et al. 2018, Pascual et al. 2018).

Lungworms (nematodes of the family Pseudaliidae) infect the respiratory tracts of cetaceans, where they may induce parasitic pneumonia, one of the most common causes of death diagnosed in porpoises stranded in the UK (Baker & Martin 1992), and are also found in the cranial sinuses and middle ear. Kuhn (1829) first described the presence of *Strongylus* (now *Pseudalius*) *inflexus* in the lungs, bronchi and typani cavity of a harbour porpoise. Lungworms typically accumulate with age in odontocetes (e.g. Raga & Balbuena 1993; see Measures 2001 for a review of lungworms in marine mammals). They are also vectors of viral and bacterial infections. Davison et al. (2010) isolated *Salmonella enterica* from lungworms *Pseudalius inflexus* removed from a harbour porpoise. It has been suggested that initial infections with the pseudaliid nematode *Stenurus minor* may stimulate protective immunity in porpoises (Faulkner et al. 1998).

The probability of finding parasites in all organs of porpoises increases with body length (ten Doeschete et al. 2017), hence presumably also with age. These authors also reported a higher probability of parasite presence in the ears and stomachs of porpoises in a poorer nutritive condition.

Nine species of parasites have been identified in harbour porpoises bycaught and stranded in Spain (Table 14), as compared to eight species from the Baltic Proper subpopulation. These numbers represent a small proportion of the 55 taxa of parasites that have been described in harbour porpoises worldwide since 1809 (Dzido et al. 2021). Some of the most common species from the global list have been detected in Galician waters, including the respiratory nematodes *Stenurus minor* and *Halocercus invaginatus*, and the gut nematodes *Anisakis simplex* (*s.s.*) and *Anisakis pegreffii*, although several other frequently recorded parasites have not yet been seen in this region, such as the liver fluke *Campula oblonga*, the gut cestode *Diphyllobothrium stemmacephalum* and the respiratory nematodes *Pseudalius inflexus* and *Torynurus convolutus*. Both larvae and adults of *Anisakis simplex* (*s.l.*) (in one case confirmed as *Anisakis simplex* (*s.s.*)) were detected in three out of four porpoises examined in Galicia between 1991 and 1996 (Abollo et al. 1998). Pons-Bordas

Table 14 Parasite Species Detected in Harbour Porpoises from Iberian Waters

Parasite Species	Prevalence	Sample Type	Year	Region	Reference
Anisakis sp.	Unknown	Stranded	2019	Galicia	Covelo and López (2021b)
Anisakis simplex (*s.l.*)[1]	66.7%	Stranded	2017–2018	Galicia	Pons-Bordas et al. (2020)
Anisakis sp.[2]	55.5% (5/9)	Stranded	2019–2021	Galicia	TRANSITION project, unpublished data
Anisakis simplex (*s.l.*)	75%	Stranded	1991–1996	Galicia	Abollo et al. (1998)
Anisakis simplex (*s.s.*)	Unknown	Stranded	2004–2019	Iberian Atlantic coast	Cipriani et al. (2022)
Anisakis pegreffii	Unknown	Stranded	2004–2019	Iberian Atlantic coast	Cipriani et al. (2022)
Anisakis simplex (*s.s.*) × *A. pegreffii* hybrid	Unknown	Stranded	2004–2019	Iberian Atlantic coast	Cipriani et al. (2022)
Crassicauda sp.	Unknown	Stranded	2019	Galicia	Covelo and López (2021b)
Halocercus invaginatus	25%	Stranded	1991–1996	Galicia	Abollo et al. (1998)
Giardia duodenalis[3]	5.9%	Stranded	2008–2012	Galicia	Reboredo-Fernández et al. (2015)
Cryptosporidium sp.	5.9%	Stranded	2008–2012	Galicia	Reboredo-Fernández et al. (2015)
Isocyamus deltobranchium	Unknown	Stranded	2007	Galicia	Martínez et al. (2008)
Toxoplasma gondii	Unknown	Stranded	Unknown	Andalusia[3]	Cabezón et al. (2004)
Stenerus minor	75%	Bycaught	2009–2018	Galicia	Saldaña Ruiz (2021)
Stenerus minor	Unknown	Stranded	2018	Galicia	Díaz Caneiro (2019)

Prevalence is specified when possible. Information about the sample type (known bycatch or stranded (the latter may also include bycaught animals)), date and region of collection is also provided. (Notes: 1. Results based on ulcerative lesions typically associated with the presence of *Anisakis simplex* s.s.; 2. Pending identification to species. 3. *Giardia duodenalis* is also sometimes listed (incorrectly) as *G intestinalis* or *G. lamblia* (Thompson & Monis 2011). 3. Andalusian Mediterranean coast.)

et al. (2020) observed ulcerative lesions associated with the presence of *Anisakis simplex* (*s.l.*) (again in one case confirmed as *Anisakis simplex* (*s.s.*)) in one of three porpoises examined which had stranded in Galicia in 2017 and 2018. Cipriani et al. (2022) identified *Anisakis simplex* (*s.s.*), *Anisakis pegreffii* and a hybrid of both species in harbour porpoises stranded along the Iberian Atlantic coast from samples collected between 2004 and 2019.

Abollo et al. (1998) recorded adult nematodes *Halocercus invaginatus* in the lungs of porpoises from Galicia. *Stenerus minor*, another nematode affecting the respiratory system and the associated cardiovascular organs, was recorded in harbour porpoises in Galician waters by Saldaña Ruiz (2021), who found between 27 and 69 specimens in each of three individuals (see also Saldaña et al. 2022), and Díaz Caneiro (2019), who counted 117 specimens in one stranded porpoise. These numbers are considerably lower than numbers reported from individual porpoises in some other regions: 2928 (Baltic Sea, Dzido et al. 2021), 8920 (Gulf of St. Lawrence, Faulkner et al. 1998) and 11,000 (Black Sea, Biserkov & Dimitrov 1991). Unlike gastrointestinal nematodes, there is a clear link between lungworm infestation and mortality, with parasitic pneumonia being a commonly reported cause of death in other regions (e.g. Neimanis et al. 2022).

Protozoan parasites such as *Toxoplasma gondii* were detected in blood antibodies in one of the few stranded harbour porpoises in the Andalusian Mediterranean coast (Cabezón et al. 2004). Ectoparasites have been observed in porpoises stranded in Galicia: the cyamid *Isocyamus deltobranchium* was present on the caudal peduncle and flukes of one individual (Martínez et al. 2008). A negative relationship between inorganic element concentrations and parasite burdens was reported for porpoises stranded along the Portuguese coast: individuals with high parasite burdens showed a low concentration of zinc and arsenic in all organ systems and nickel in the renal system; it was also noted that such animals tended to be those with better nutritional condition (Ferreira et al. 2016). This may simply imply that animals eating more tend to ingest more parasites.

Other anthropogenic threats

Human activities at sea can cause mortality of porpoises, e.g., due to ship strikes, as well as having sub-lethal effects due to habitat degradation and disturbance, including that caused by underwater noise. Ship strike mortality is perhaps best known in the context of large cetaceans but it also affects small cetaceans and many other marine species. A review by Schoeman et al. (2020) cites published reports of ship strike mortality of harbour porpoises in the UK, the Netherlands and Canada. European marine waters in general have a high density of shipping, with the Bay of Biscay and Iberian coast being one of the areas that accommodates major shipping routes (e.g. Kinneging 2022).

Underwater noise and disturbance

The effect of underwater noise on cetaceans and other marine life has attracted considerable attention (e.g. Thomsen et al. 2021). To date, there has been no published assessment of underwater noise for the Iberian Peninsula (Kinneging 2022) although there have been studies of specific areas, e.g., the Mediterranean port of Cartagena (Rodrigo et al. 2022). In the USA, a framework was developed to assess the Population Consequences of Acoustic Disturbance (PCAD) for marine mammals (NRC 2005), later further developed as more general framework (Population Consequences of Disturbance (PCoD)) (Pirotta et al. 2018). One of the few examples of its application in Europe was a study of the effects of offshore windfarm development in the North Sea on porpoises (King et al. 2015). Avoidance of vessels by porpoises is well known and hence is taken into account when estimating abundance from boat-based surveys (e.g. Palka & Hammond 2001), and there is evidence from both wild and captive studies that porpoises respond to vessel noise. Dyndo et al. (2015) reported that porpoises responded to the noise from passing vessels by displaying stereotypical "porpoising" behaviour, noting that, in the wild, this would not only have an energetic cost (investment in moving, loss of foraging opportunities) but could also lead to abandonment of calves. Wisniewska et al. (2018) demonstrated that vessel noise disrupts foraging in porpoises.

As acoustic deterrents (pingers) on nets have been increasingly deployed in attempts to reduce fishery bycatch of porpoises, the possible effects of the noise they produce, in terms of disruption of behaviour, displacement from foraging areas, stress, hearing loss, and impacts on individual health and reproductive output, and population viability, have also received attention, based on field observations, experimental studies and modelling (e.g. Culik et al. 2001, Kastelein et al. 2000b, Lusseau et al. 2023). This topic is revisited in section ‘Current status, knowledge gaps and future research, monitoring and conservation’.

Given the high energetic requirements of the Atlantic harbour porpoises, and hence the large proportion of time they need to spend feeding, it is expected that they will present low resilience to disturbance (Wisniewska et al. 2016), especially if displaced from their feeding grounds due to, for example, continuous underwater sound (IAMMWG et al. 2015). The presence of motor boats and fishing vessels was found to negatively affect the presence and density of porpoises recorded off Galicia (Díaz López & Methion 2018). Similar results were found in central Portugal, where the porpoise sighting rate increased as the number of fishing boats decreased (Pereira 2015). Further work is needed to understand whether disturbance from vessel traffic represents a population-level threat to Iberian porpoises.

Other anthropogenic sources of disturbance documented in other parts of the world but not in the Iberian Peninsula include offshore renewable energy development (e.g. Simmonds & Brown 2010) and seal deterrents used at fish farms (Findlay et al. 2021). Effects may include not only increased stress and displacement from preferred feeding areas but also, in the case of Acoustic Deterrent Devices (ADDs) used in aquaculture, auditory impairment. Despite the increasing importance of aquaculture in Spain, notably in Galicia where species such as turbot, sea bream and mussels are cultured, seals are occasional (vagrant) visitors to most of the Iberian Peninsula (with the exception of a very small resident population of Mediterranean monk *Monachus monachus* seal off Madeira), and we are not aware of the use of acoustic deterrents. The development of offshore renewable energy production has the potential to represent an additional pressure in the future.

Prey depletion

Porpoises are also thus likely to be susceptible to prey depletion due to their high feeding rate (Wisniewska et al. 2016). MacLeod et al. (2007) highlighted the relatively high frequencies of deaths attributed to starvation in porpoises along the Scottish coast at times of low sandeel abundance, although the existence of any causal relationship between the two variables remains speculative: it is sometimes difficult to determine whether weight loss in such animals was due to low food availability or some underlying pathology. Nevertheless, the incidence of starvation/emaciation has been increasing, among necropsied porpoises in UK waters (Deaville et al. 2018), and it was also a leading cause of death among porpoises that died during an unusual mortality event in Dutch waters in 2011 (IJsseldijk et al. 2022).

The main prey of porpoises in Galicia (NWIP) include blue whiting, *Trisopterus*, hake and scad, all of which are commercially important, and at least in terms of numbers eaten, gobies and silvery pout, which are not (Pierce et al. 2010, Read et al. 2013, Hernandez Gonzalez et al. 2024); “see also Feeding ecology”. Stocks of scad, European hake and blue whiting are all assessed by ICES. Two stocks of scad occur in the area (see Brunel et al. 2016), one on the north coast (the western stock), which is currently at a historically low level (ICES 2018b), and one on the west coast (the southern stock) which, while less abundant, appears to be increasing (ICES 2018b). The southern stock of European hake is distributed in the Cantabrian Sea and Atlantic Iberian waters while blue whiting forms a single wide-ranging stock, the distribution of which includes the Iberian Peninsula. The abundance of both these stocks is currently above MSY (ICES 2018b). The decline of the western stock of scad in ICES subarea 8 (including the north Spanish coast) is a potential cause for concern. Despite its commercial importance, *Trisopterus luscus* is not assessed by ICES. In general terms, prey depletion is probably not an issue for Iberian porpoise, although low availability of energetically rich species such as sardine and scad could have adverse consequences. The amount of some commercial fish species removed by porpoises was estimated and compared with removal by fisheries (Santos et al. 2014), suggesting that porpoises removed 1% or less of the amount of hake taken by fisheries and less than 0.1% of the amount of sardine taken by fisheries hake. The abundance estimate for Iberian porpoise available at the time was around 1100 porpoises, subsequently revised to approximately 2900, but the impact of porpoises on fishery catches is still likely to be minimal. However, the impact of competition with fisheries on Iberian harbour porpoises is unknown.

Interactions with other species

In the North Atlantic, harbour porpoises are known to be preyed upon by killer whales *Orcinus orca* (e.g. Samarra et al. 2018), grey seals *Halichoerus grypus* (Leopold et al. 2015) and large sharks, e.g., great white shark *Carcharodon carcharias* (Arnold 1972) and Greenland shark *Somniosus microcephalus* (Williamson 2011). The Atlantic coast of Spain is within the known distribution of great white shark, although records of its occurrence in Galicia are considered questionable (the records are of fish that were likely caught elsewhere) (Bañón et al. 2010). According to Fishbase, records of Greenland shark in the eastern Atlantic are rare further south than Ireland and the species is not known from Galician waters (Bañón et al. 2010). Orcas are resident in the Strait of Gibraltar and are sometimes seen along the northwest coast of Spain –– they have recently attracted attention due to interactions with small boats (Esteban et al. 2022). However, this subpopulation of orcas appears to specialise in feeding on tuna (García Tiscar 2010).

Fatal interactions between porpoises and dolphins are well known, mainly involving bottlenose dolphin. Lethal attacks on harbour porpoises by bottlenose dolphins are most common where there is a resident population of the latter, e.g., in northeast Scotland where the phenomenon was described by Ross and Wilson (1996) and remains among the most commonly diagnosed cause of death in porpoises in Scotland (e.g. Davison & ten Doeschate 2020). It has been suggested that such attacks are a result of competition for food resources or that male bottlenose dolphins are practising infanticide (male bottlenose dolphins are known to kill the offspring of other males). Such lethal interactions may also have indirect consequences, including the avoidance by porpoises of areas utilised by bottlenose dolphins. MacLeod et al. (2007) proposed that, where porpoise distribution overlaps with bottlenose dolphin distribution, there is selective pressure for low fat reserves in porpoises because it improves their flight performance and hence increases their ability to avoid fatal bottlenose dolphin attacks.

Although apparently relatively rare, mortalities of porpoises due to bottlenose dolphin aggression are thought to occur in the NWIP (Pierce et al. 2010, López-Fernández & Martínez-Cedeira 2011). No direct observation has been reported to date, but evidence based on examination at necropsy of fresh carcasses shows that fatal interactions with bottlenose dolphins are most likely occurring in Galicia (i.e. presence of rake marks from bottlenose dolphin teeth on the porpoise's skin, fractured scapula and ribs, subcutaneous hematoma) (López & Rodríguez 1995, Alonso et al. 2000). Avoidance behaviour has been observed in common dolphins in the presence of bottlenose dolphins in Ría de Arousa (Galicia), and a fatal attack by a bottlenose dolphin on a solitary common dolphin was reported in the same area (Methion & Díaz-López 2021).

Given that porpoise and bottlenose dolphin distributions also overlap in Portugal, it is likely that such interactions also occur in Portugal. An interspecific interaction was suggested as the possible origin of the trauma that caused the death of the leucistic porpoise had been observed in the Douro River by Gil et al. (2019), which stranded in August 2020 (A. Torres-Pereira, Pers. Comm.).

Conservation of porpoises in Europe: legal protection and its implementation

International agreements

The harbour porpoise is covered by several international agreements, including the Convention of Migratory Species (CMS) and its associated agreements, and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), as well as European Directives and national legislation. The harbour porpoise is also listed on the IUCN Red List: at global level, the species is of Least Concern (Braulik et al. 2020); in Europe as a whole, its status was updated from Vulnerable to Least Concern in 2023 (Temple & Terry 2007, Sharpe & Berggren 2023); and it is Critically Endangered in the Baltic Sea (Hammond et al. 2008) and Endangered in the Black Sea (Birkun & Frantzis 2008). There is no separate entry for the Iberian porpoise (see Carlén et al. 2021 for a review).

In northern Europe, ASCOBANS, which derives from the CMS and is administered by the United Nations Environment Programme (UNEP), promotes the conservation of small cetaceans and requires Parties to undertake various monitoring and conservation actions. Spain and Portugal are Range States (their seas fall within the area of the Agreement) but neither is a Party to the Agreement. A second CMS-derived agreement, ACCOBAMS, fulfils a similar role for southern Europe. Its current range extends to the Iberian Peninsula and thus overlaps with that of ASCOBANS; both Spain and Portugal are Contracting Parties. ASCOBANS has “long recognized bycatch as the most significant threat to the small cetacean populations in the Agreement Area and resolutions addressing this problem have been passed repeatedly at Meetings of the Parties. These resolutions address the monitoring programmes and mitigation measures required in order to understand the situation and reduce mortality of cetaceans”.[5] While ACCOBAMS has arguably placed less emphasis on bycatch in its conservation actions, the Agreement “provides that Parties shall apply, within the limits of their sovereignty and/or jurisdiction and in accordance with their international obligations, appropriate measures for the assessment and management of human-cetacean interactions, stressing that measures concerning fisheries activities shall be applied in all waters under their sovereignty and/or jurisdiction, and outside these waters in respect of any vessel under their flag or registered within their territory”.[6] ACCOBAMS and ASCOBANS established a Joint Bycatch Working Group in 2019 and co-organised a workshop on Current cetacean bycatch issues in European waters at the 34th Annual Conference of the European cetacean Society in April 2023.

The systematic recording of strandings in Europe dates from the early twentieth century. The Natural History Museum in the UK began systematic recording of strandings in 1913 and strandings monitoring in France started in 1971. Strandings have been recorded in Galicia since 1973, originally by the Sociedade Galega de Historia Natural. For larger cetaceans such as the sperm whale, historical stranding records going back several centuries can be reconstructed (e.g. Smeenk 1997). Several countries and regions initiated or upgraded strandings monitoring at the start of the 1990s, reflecting commitments formalised under the ASCOBANS Agreement[7] in 1992. The Agreement included a commitment to establish an efficient system for reporting stranded animals; to carry out necropsies to determine the cause of death, study feeding habits and collect tissues for further studies; and to make the information available in an international database. Spain (in 1999) and Portugal (in 2005) joined ACCOBAMS rather than ASCOBANS, but the current Galician

[5] https://www.ascobans.org/en/species/threats/bycatch

[6] https://accobams.org/conservations-action/bycatch-depredation/

[7] https://www.ascobans.org/sites/default/files/basic_page_documents/Ch_XXVII_09_CertifiedTrueCopiesAgreement

strandings network, operated by CEMMA, started in 1990, and the network in northern Portugal dates from 2000.

The 3rd Meeting of Parties of ASCOBANS in 2000 (see ASCOBANS 2000) passed a resolution (Resolution 3.3[8]) about the incidental take of small cetaceans, which noted that "the aim of ASCOBANS can be interpreted as 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" and "the general aim should be to minimise (i.e. to ultimately reduce to zero) anthropogenic removals within some yet-to-be-specified time frame, and that intermediate target levels should be set". 'Unacceptable interactions' were defined as being "in the short term, a total anthropogenic removal above 1.7% of the best available estimate of abundance" and there was an "intermediate precautionary objective to reduce by-catches to less than 1% of the best available population estimate". It was also noted that "if available evidence suggests that a population is severely reduced", then "an anthropogenic removal of much less than 1.7%" could be considered as an "unacceptable interaction".

Applied to the best estimate for the population size of Iberian porpoise (2900), the 1.7% limit suggests an annual removal limit of 49 animals and 1% would be 29 animals. At its COP 13 meeting in 2020, noting the importance of reducing unintentional mortality from fisheries bycatch, under the aforementioned Concerted Action for Baltic and Iberian harbour porpoises, CMS (2020) proposed "developing an action plan for the Iberian harbour porpoise population" before COP 14 (which was scheduled to be held in October 2023). ASCOBANS (2021) developed its aforementioned draft proposal for inclusion of the Iberian porpoise in the Appendices of CMS, citing the benefits to the population by listing it in Appendices I (leading to a collaborative transboundary management plan) and II (offering strict Range State protections, especially in relation to reducing unintentional mortality from fisheries bycatch). During 2023, ASCOBANS held a workshop to recommend new conservation objectives for small cetaceans in relation to anthropogenic removals. Although the outcomes from this workshop are as yet unpublished, it was agreed that the 1% and 1.7% limits were now considered inappropriate as they "would not achieve the desired outcome either when applied to the harbour porpoise or for other species".

European Union Environmental Directives

The EU Habitats Directive (Council Directive 92/43/EEC 1992), which implemented the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention), and the Marine Strategy Framework Directive (MSFD; Directive 2008/56/EC 2008) have both resulted in increasing efforts to monitor the status of cetacean populations and to implement conservation measures. Since EU Member States report on the conservation status of cetaceans under both the Habitats Directive and MSFD, the status of porpoises has been assessed at national level. The realisation that the Iberian porpoises represent a genetically isolated population and potentially a distinct subspecies adds urgency to the need for a separate assessment of this population. ICES WGMME (ICES 2013c) recommended that the Iberian population of porpoises should be treated as a separate Management Unit. In practice, the porpoises found in Spanish and Portuguese waters are likely to belong almost exclusively to the Iberian population. Note, however, that Ben Chehida et al. (2023) referred to two porpoises from southern Portugal "carrying a divergent haplotype closely related to those from the Mauritanian population".

The Habitats Directive protects all cetaceans from disturbance and incidental killing and capture, while harbour porpoise is one of two species for which the designation of Special Areas of Conservation (SACs) was proposed. Several European countries have designated SACs for harbour porpoise, including the UK and Ireland. SACs are one of several types of Marine Protected Area falling under the Natura 2000 umbrella (which also includes terrestrial sites). The European Environment

[8] https://www.ascobans.org/sites/default/files/document/MOP3_2000-3_IncidentalTake_1.pdf

Agency maintains a publicly accessible EU database on Natura 2000 that is updated annually, including an online map viewer and a visualisation of area coverage and site numbers by Member State[9,10].

Spain has numerous Natura 2000 marine (or partly marine) sites linked to the Habitats and/or Birds Directives along its Atlantic and Mediterranean mainland coasts, in its north African enclaves and in the Canary Islands. Two SACs (Zonas Especial de Conservación) have been created by Royal Decree, Estrecho Oriental (located to the east of the Strait of Gibraltar; Real Decreto 1620/2012, de 30 de noviembre) and Islas Chafarinas (off the coast of Morocco, to the east of the enclave of Melilla; Real Decreto 190/2018, de 6 de abril). Harbour porpoise is mentioned in the first of these Royal Decrees although in the online map viewer porpoises are not listed under either site. According to the Marine Conservation Institute's Marine Protection Atlas[11], 13% of Spain's total marine area of 1,008,165 km^2 is under some designated protection (10% implemented, 3% unimplemented) although less than 1% of the marine area is highly or fully protected from fishing.

The Spanish Natura 2000 sites that can be viewed in the online map viewer, for which harbour porpoise is included in the associated list of species, are almost all in Galicia and Asturias, on the northwest and north Atlantic coasts. Galicia has nine sites: Complejo Ons-O Grove (7607 ha, i.e., 76 km^2, Habitats Directive), Complexo intermareal Umia – O Grove, A Lanzada, punta Carreirón e lagoa Bodeira (2813 ha, Birds Directive), Costa da Mariña occidental (2169 ha, Birds Directive), Costa da Morte (11,809 ha, Habitats Directive), Costa da Morte (Norte) (7962 ha, Birds Directive), Costa da Vela (1418 ha, Habitats Directive), Illa de Ons (924 ha, Birds Directive), Illas Cies (990 ha, Habitats and Birds Directives) and Illas Estelas (725 ha, Habitats Directives). Asturias has five sites: two named Cabo Busto-Luanco (11,642 ha, Habitats Directive, and 10,043 ha, Bird Directive), Penarrondo-Barayo (4317 ha, Habitats and Birds Directives), Sistema de cañones submarinos de Avilés (338,961 ha, Habitats Directive) and Yacimientos de Icnitas (3560 ha, Asturias, Habitats Directive). The remaining site is Doñana in Andalucia (128,268 ha, Habitats and Birds Directives), a National Park comprising mainly of terrestrial wetlands plus a small area of coastal waters[12].

Portugal has designated 4.5% of its 1,728,718 km^2 marine area, of which less than 1% is highly or fully protected from fishing. Portugal has three Natura 2000 marine sites for which habour porpoise is listed, all designated under the Habitats Directive, although none is an SAC: Maceda – Praia da Vieira to the south of Porto (502,637 ha), Arrábida/Espichel (20433) south of Lisbon and Costa Sudoeste (262,299 ha) in the Algarve[13].

It is apparent that only a small proportion of Iberian Atlantic coastal waters is covered and that, generally, the process of designating sites as SACs has not been completed. In fact, Carlén et al. (2021) argued that, in reality, SACs offered very limited protection (see also Pinn et al. 2021). To date, the Habitats Directive has not demonstrably contributed to the reduction of cetacean bycatch. Firstly, the requirement to reduce incidental killing under this Directive has essentially been ignored until recently with, arguably, too much focus to the establishment of SACs. Secondly, the EC has been slow to challenge Member States in relation to their failure to deliver the required SACs. It can also be argued that declaration of an SAC could benefit a species that has small local resident populations, like bottlenose dolphin, and the benefit for more wide ranging species like harbour porpoise is less obvious.

Article 17 of the Habitats Directive requires Member States to report every 6 years about the progress made with implementing the Habitats Directive. Since the main objective of the Directive is to maintain or restore Favourable Conservation Status for habitats and species of Community interest, reporting under Article 17 is focused on the status and trends of these habitat types and species. Reports for the 2013–2018 period were issued in 2019. In relation to assessment under Article 17 of the Habitats

[9] https://www.eea.europa.eu/themes/biodiversity/natura-2000/the-natura-2000-protected-areas-network
[10] https://natura2000.eea.europa.eu/
[11] https://mpatlas.org/
[12] https://mpatlas.org/countries/ESP/
[13] https://mpatlas.org/countries/PRT/

Directive, Spain assessed the Conservation Status of porpoises to be "Unfavourable-Inadequate" for the period 2013–2018, while Portugal assessed it as "Unfavourable-Bad". For the previous assessment (2007–2012), both countries had assessed it as "Unfavourable-Inadequate", while for 2001–2006, Portugal had rated it as "Unfavourable-Inadequate" and Spain had assessed it as "Unknown".

The MSFD aims to effectively protect the marine environment across Europe, including the marine resource base upon which human economic and social activities depend, through the achievement of Good Environmental Status (GES) of EU waters. Eleven Descriptors of GES have been defined, the first of which is that "Biodiversity is maintained". The criteria to assess the status of cetacean populations under the MSFD in relation to Descriptor 1 (biodiversity) include abundance, distribution, demographic characteristics and the bycatch mortality rate (e.g. Palialexis et al. 2019). Reporting under the MSFD follows a 6-year cycle, the first cycle being completed in 2018. For the first cycle, both Spain and Portugal reported on the status of harbour porpoise in relation to several Criteria for Descriptor 1 (Table 15). While different indicators provided different outcomes, and in most cases, the quantity and/or quality of evidence available was a limiting factor, the overall species assessment in the First Cycle Reports for Spain and Portugal was "Not good".

The Habitats Directive and MSFD do not include specific instructions about how their requirements should be implemented, and the conservation objectives are not quantitatively defined. Developing the necessary practical framework (e.g. the indicators, criteria, baselines, thresholds, assessments, monitoring programmes and programmes of measures) and associated infrastructure has fallen to the Member States, with assistance of European, intergovernmental and international bodies such as the Joint Research Centre (JRC), the International Council for the Exploration of the Sea (ICES) and the (European) Regional Sea Conventions: OSPAR (North East Atlantic), HELCOM (Baltic Sea), Barcelona Convention (Mediterranean) and Bucharest Convention (Black Sea). The EU has also funded numerous projects that aimed to advance and improve the implementation of these directives, notably under the LIFE and MSFD programmes.

In the context of assessing the status of cetaceans in the OSPAR Maritime Area, the OSPAR Marine Mammal Expert Group (OMMEG) has proposed guidelines to set thresholds for anthropic removals, including bycatch, aligning with the conservation objective of maintaining or restoring population to 80% of carrying capacity with probability 0.8 over 100 years (Taylor et al. 2022). These were agreed at the OSPAR Biodiversity Committee meeting in autumn 2021. Given current abundance data on Iberian harbour porpoise, a removal limit of zero should be applied to the Iberian porpoise population (see ICES 2021a, Taylor et al. 2022). This was based on a modified PBR approach (*m*PBR, modified because the underlying conservation objective differs from the original conservation objective of the US Marine Mammal Protection Act), applied to a minimum estimate of population size (N_{min}) and defined as the 20th percentile of the log-normal distribution. If N_{min} was less than 2500 mature animals, *m*PBR was set to zero. Using the Hammond et al. (2021), estimate of the population size for Iberian porpoise, 2898 (CV = 32%) animals, results in an N_{min} value

Table 15 Assessment by Portugal and Spain of the Status of Harbour Porpoise under Descriptor 1 of the MSFD for Each Criterion in the First Cycle Reports

Criterion	Description	Portugal	Spain (North)
D1C1	Bycatch mortality	Not good	Not good
D1C2	Abundance	Not good	Good
D1C3	Demographic characteristics	Not assessed	Not assessed
D1C4	Distribution range and pattern	Good	Good
D1C5	Habitat	Not good	Not assessed
Species assessment		Not good	Not good

Source: Adapted from Anon. (2022).

Porpoise was not assessed for southern Spain.

of 2122, hence a bycatch limit of zero. As previously noted, ICES (2021a) estimated marine mammal bycatch mortality in the OSPAR region but were unable to provide an assessment for Iberian harbour porpoise due to inadequate data on the number of bycaught animals.

It is worth noting that the aspirational approach of European Union conservation Directives concerning conservation contrasts with the detailed prescriptive approach evident in legislation from the USA, for example, the Marine Mammal Protection Act (see Verutes 2022 for a critical review). Having said this, the MMPA remained ineffective until 1994 when PBR was introduced. Previously, management action was to be triggered by the population size falling below the maximum net productivity level (MNPL, similar conceptually and mathematically to Maxium Sustainable Yield (MSY) in fisheries) but, as observed by Taylor et al. (2000), it "failed as a management tool because the parameter that triggered management action was extremely difficult to estimate for the majority of populations". In Europe, it has been necessary not only to develop the management rules but also to define the quantitative conservation objective around which the rules should be designed. Arguably, as mentioned above, there has also been a failure of enforcement by the EC. Furthermore, fishery bycatch of protected species in the EU is covered by both environmental and fisheries law and the overlap in competences and discrepancies in objectives need to be addressed (see, e.g., Fock 2011, Pinn et al. 2021).

European Fisheries Regulations

In addition to the EU's "conservation" Directives, legislation aimed at monitoring and reducing fishery bycatch mortality of cetaceans is also relevant for porpoises. The European Union's Common Fishery Policy originated in 1970 and was reformed in 2002 and 2013, with the latter reform including an increased emphasis on sustainability (e.g. Orach et al. 2017). Monitoring and mitigation of cetacean bycatch was specifically addressed by Council Regulation (EC) No. 812/2004, which mandated the deployment of dedicated observers on-board a sample of vessels in certain fishing fleets to monitor cetacean bycatch, as well as the deployment of acoustic deterrent devices, known as "pingers", on fixed nets in some areas (Kraus et al. 1997, Culik et al. 2001, Pinn 2023). The Regulation stated: "The use of such devices should therefore be required in areas and fisheries with known or foreseeable high levels of by-catch of small cetaceans, and taking into account the cost/efficiency of such requirement". The regulation also encouraged pilot projects and scientific studies on the effects of using pingers. Under the general monitoring obligations of the Regulation, on-board observer coverage aimed to achieve a Coefficient of Variation of 0.30 or less for the estimate of bycatch per unit effort for the most frequently caught species. For pilot projects, minimum observer coverage was also specified, as 10% of fishing effort by pelagic trawlers, covering at least 3 vessels, and 5% or at least 20 vessels for other gears.

As was also the case for several other EU Member States, Spain and Portugal were arguably slow to respond to Regulation 812/2004. Thus, Spain undertook a single Pilot Study and used fishery observers (not dedicated observers) to record cetacean bycatch. In general, across Europe, the success of the Regulation was limited by the fact that it did not apply to all relevant fleets, monitoring intensity was low where it did apply, fisher participation was essentially voluntary and there was little or no monitoring or enforcement of its implementation (e.g. Dolman et al. 2016, 2021, Read et al. 2017).

Regulation 812/2004 was repealed and replaced in 2019 by the Regulation on the Conservation of Fishery Resources and the Protection of Marine Ecosystems through Technical Measures (2019/1241). This offered some advances, notably including an explicit obligation to ensure that bycatch of sensitive species is "minimised and where possible eliminated", a requirement for relevant technical measures to be applied regionally in high risk fisheries, and a requirement for Member States to report on the effectiveness of monitoring of bycatch and measures taken to reduce it. However, the regulation did not clearly define target thresholds for the implementation of mitigation, and the level of monitoring being undertaken was (and is) insufficient to obtain the robust estimates of bycatch rate needed to determine the impact of bycatch at population level (Dolman et al. 2021).

The European Commission adopted the Marine Action Plan in February 2023. The Plan contributes to delivering on the EU Biodiversity Strategy 2030, and under heading "Action to improve fishing selectivity and reduce the impact of fishing on sensitive species", "The Commission calls on Member States to…:

> Adopt national measures or joint recommendations to the Commission to minimise by-catch (or reduce it to the level that enables the full recovery of the populations) of: by the end of 2023, harbour porpoises in the Baltic Proper and the Black Sea, the Iberian Atlantic and the common dolphin in the Bay of Biscay…[14]

National legislation

Spain

Real Decreto 1727/2007, de 21 de diciembre[15], introduced various protection measures for cetaceans in Spain. The Spanish Catalogue of Threatened Species listed the harbour porpoise as "vulnerable" in 2011[16], and this was updated in 2020 to reflect a change in status to "in danger of extinction"[17].

The Ministerial Order APA/1200/2020 of 16th December, published on 18th December 2020 in the Official State Bulletin (Boletín Oficial del Estado, BOE), established new measures for mitigation and monitoring to improve scientific knowledge to reduce cetacean bycatch on fishing activities. Its provisions were described by ICES (2023) and include the following:

- Article 3 of APA/1200/1200 established an on-board observer programme focused on cetacean bycatch. All fishing vessels operating in national fishing grounds of the Bay of Biscay and the Iberian coast, as well as in non-Spanish European waters in the Bay of Biscay, must take scientific observers on board when requested to do so by the General Secretariat for Fisheries. The programme is focused on the national fleet segment that, according to scientific analysis, poses the highest risk of interaction with vulnerable species. The aim is to cover at least those trawling activities involving a major vertical opening (pair bottom trawl), as well as vessels using bottom gillnets or trammel nets with a mesh size equal to or bigger than 80 mm. Article 3 also proposed to complement the observer programme with a pilot project on remote electronic monitoring (REM). REM systems have automatic sensors that cannot be manipulated and, as such, should provide reliable records of fishing operations. Currently, only REM systems with associated image analysis software are used. Since they are connected to on-board navigation systems, these images are linked to specific stages of the fishing process. Associated software analysis makes the image processing more efficient and ensures stronger and more reliable data, excluding the possibility of human error.
- Article 4 of APA/1200/2020 established the obligation to use acoustic deterrent devices (pingers) for all Spanish bottom trawlers whose fishing activity is conducted in Cantabrian and Northwest fishing grounds in national waters and in the non-Spanish EU waters of the Bay of Biscay. Currently, 65 trawlers are using pingers in the North Western Cantabrian Sea, plus 12 more in the non-Spanish EU waters of the Bay of Biscay. The devices are used

[14] https://faolex.fao.org/docs/pdf/eur217860.pdf
[15] https://www.boe.es/boe/dias/2008/01/12/pdfs/A02292-02296.pdf
[16] https://www.boe.es/eli/es/rd/2011/02/04/139
[17] https://www.boe.es/eli/es/o/2020/11/20/ted1126

in accordance with the Annex to Commission Implementing Regulation (EU) 2020/967 of 03.07.2020, which lays down rules on the signal and implementation characteristics of acoustic deterrent devices as described in Part A of Annex XIII of Regulation (EU) 2019/1241 of the European Parliament and the Council on the conservation of fisheries resources and the protection of marine ecosystems through technical measures.
- Article 6 established a move-on rule for fishing activities using bottom trawl gear. If more than three cetaceans are caught in the same fishing manoeuvre, or any cetacean is caught in two consecutive hauls, fishing vessels shall move a minimum of 5 miles from the relevant point to continue their fishing activities, at a high navigational speed.
- Article 7 of APA/1200/2020 established the obligation to notify by-catch in logbooks. All fishing vessels, irrespective of the gear used, are obliged to record and transmit information on all bycatch events involving any cetacean species, via their logbook. They should indicate the number of specimens caught, the species, their vital status and relevant morphological characteristics, such as approximate size or whether they show previous marks of possible contact with fishing gear.

On 8th March 2021, the BOE published the Resolution of 2nd March, by the General Secretariat for Fisheries, allocating fishing quotas for scientific purposes in order to implement pilot projects on REM in the context of the mitigation measures for cetacean bycatch. The first stage of this initiative included 13 vessels using bottom trawling and gillnetting. The Resolution of the 15th March 2022, by the General Secretariat for Fisheries provided the basis for continuing the REM project, involving 21 vessels: 61.5% more than those involved in 2021. The project is expected to continue in 2023.

Portugal

Portugal was one of the first European nations to protect cetaceans when it issued Decree-law nº263/1981 of 3rd September. The decree approved the Regulation for the Protection of Marine Mammals in Inland Waters, the Territorial Sea and the Portuguese Continental Exclusive Economic Zone, recognising the high scientific value of cetaceans, the rarity of some species in the seas of mainland Portugal, the alarming decline in the populations of others and the need to start to take conservation measures to protect these populations. Article 2 of this Decree established that it is expressly forbidden, throughout the year, to fish for, catch or kill any species of marine mammal that may occur in estuaries and the Continental Exclusive Economic Zone of Portugal. Article 3 prohibits the sale and commercialisation of marine mammals in fish auctions, markets or anywhere else (even those found dead in nets, fishing gear or on land). Article 4 obliges specialised scientific institutions to assist marine mammals stranded alive. After any necessary assistance, they must be returned to their natural environment as soon as possible. Article 5 of Decree-Law no. 263/1981 established that infringements of the provisions of Articles 2 and 3 would be punishable by seizure and fines of 100,000 escudos (~ €500) for seals, dolphins or porpoises and 900,000 escudos (~ €4500) for sperm whales, rorquals or whales (charged per specimen).

The Decree-Law 140/99 of 24th April[18], later amended by Decree-Law 49/2005 of 24th February, transposed into national law the regulations from EU Habitats Directive (Council Directive 92/43/EEC 1992). Since 2004, following EU Regulation 812/2004, Portugal began using observers to record bycaught cetaceans, using fishery observers (already in place under the EU Data Collection Framework) as well as dedicated observers working on specific projects. Decree-Law no. 9/2006 of 6th January[19] established responsible behaviour during cetacean watching activities on the part of tour operators or audio-visual recorders, sea sports persons, researchers and the general public. Under this regulation, tour operators need prior authorisation to observe cetaceans.

[18] https://faolex.fao.org/docs/pdf/por22472.pdf
[19] https://diariodarepublica.pt/dr/detalhe/decreto-lei/9-2006-168231

The two most recent regulations published are the Ministerial Order 172/2017 of 25th May[20] and the Normative Order no. 19/DG/2020[21] issued by the Portuguese Directorate-General for Natural Resources, Safety and Maritime Services (DGRM).

Article 5 of the Ministerial Order 172/2017 of 25th May stated that (1) Nets used for fishing with traditional beach seines must have acoustic deterrent equipment installed to prevent by-catches of marine mammals, such as harbour porpoises or bottlenose dolphins; (2) If existing data indicate that fishing by traditional beach seines has no impact on cetacean populations, vessels operating in certain zones may be excluded from the first obligation; and (3) the characteristics of the equipment shall be determined by the Order of the DGRM.

Normative Order no. 19/DG/2020 from DGRM drew on the results of the Life+ MARPRO project to set rules for the deployment of pingers. Thus, the use of pingers on traditional beach seines nets is considered unnecessary to protect harbour porpoise and common dolphin but not bottlenose dolphin. Also the lack of reported interactions between marine mammals and the traditional beach seines to the south of Praia da Vieira suggests that the use of pingers in this area is not advised (which will avoid the unnecessary creation of noise pollution). The technical characteristics of the acoustic deterrent equipment were also specified (e.g. in terms of the frequency and pressure level of the emitted sounds (10 kHz and 132 dB for porpoises), the type of battery and the need for it to have a sufficient charge level), and it was specified that one pinger must be installed in each boat and one at the mouth of the net for the duration of the fishing operation and that pingers had to be installed and active in fishing operations within 10 days of the effective date of the order (i.e. by 16 August 2020). Three years on, there has been no routine enforcement, monitoring or assessment of the efficacy of this normative order, although there has been some evaluation of progress, in collaboration with fishers, in the context of two scientific projects (iNOVPESCA and CetAMBICion) in the south of Portugal.

Portaria 201/2019 of 28th June established the management plan for the Sítio de Importância Comunitária (SIC, Site of Community Importance) Maceda-Praia da Vieira, an important area for harbour porpoises. Associated actions include the existing stranding network, a fully equipped rehabilitation centre, abundance surveys and delivery of pingers to fishers, although these have been driven by other obligations; new action has been minimal.

The Red Book of Portuguese Vertebrates assessed the harbour porpoise as "Vulnerable" in 2005 (Cabral et al. 2006) and recently updated this assessment to "Critically Endangered" in the Red Book of Portuguese Mammals (Torres-Pereira et al. 2023b).

Current status, knowledge gaps and future research, monitoring and conservation

Distribution and abundance

Current situation

Information on distribution derives from abundance surveys, opportunistic sightings, strandings and genetic studies. Porpoises seem to be most common on the west coast of the Iberian Peninsula, along the Galician and Portuguese coasts. Almost all sightings are from continental shelf waters and the species is a rare visitor to the Macaronesian islands. There is genetic evidence of past (possibly continuing) emigration into the Celtic Sea. There is insufficient evidence to determine whether the distribution range is expanding or contracting.

The population size of the Iberian porpoise population appears be 3000–4000 animals. Estimates from the SCANS-II (2005) and SCANS-III (2016) surveys were almost identical at

[20] https://dre.tretas.org/dre/2982137/portaria-172-2017-de-25-de-maio

[21] https://www.dgrm.pt/documents/20143/46478/Despacho+19_DG_2020.pdf/6a1bb004-127e-61a4-1adb-257a1225c766

2880 (CV = 0.72) and 2900 (CV = 0.32), respectively, while the estimate from SCANS-IV (2022) was 4034 (CV = 0.35); in all cases, the 95% confidence intervals were wide (e.g. 1842–7309 for SCANS-IV), and the change in the estimate between 2016 and 2022 cannot be assumed to represent a real increase in abundance, although the distribution of sightings was rather different in 2022 (Gilles et al. 2023). The highest published estimate from national surveys was over 3500 (for Portugal) but this is within the 95% CI of the SCANS estimates. This is evidently a small population, and while the large-scale surveys indicate no change in abundance over 11 years, the wide confidence intervals do not preclude upward or downward trends. The population might extend offshore in Portuguese waters beyond the SCANS survey area and the large-scale surveys excluded the Galician rías but it seems unlikely that this has resulted in a serious underestimate. Although the annual abundance estimates for Portugal in 2011–2015 reported by Torres-Pereira et al. (2022) varied considerably between years, they were all arguably broadly consistent with the SCANS results. The low number of porpoise sightings during most surveys (reflecting the low density of porpoise occurrence) is a limitation. Evidence from strandings suggested a decline in porpoise abundance in Portugal in the 1980s. Genetic (Mt DNA) data indicate a loss of genetic diversity (as well as outward movement of animals into the Bay of Biscay and Celtic Sea), and although the research is ongoing, the available genetic results support the idea of a declining population (Ben Chehida et al. 2023).

Future research and monitoring

There remains a need for more frequent large-scale surveys. SCANS IV took place in summer 2022, reducing the time-gap between surveys to 6 years but (assuming this frequency can be maintained) this still means there is only one large-scale survey per reporting cycle for the Habitats Directive and the MSFD.

It would be useful to have better information on movements of porpoises within (and out of) the Iberian Peninsula. Regular surveys of distribution and abundance covering all seasons would be useful, extending into both the Galician rías and offshore waters. Although there are few offshore sightings of porpoises in the Iberian Peninsula (e.g. one mentioned by Aguilar et al. 1983), few offshore surveys have specifically looked for porpoises, and the lack of sightings may be partly due to the difficulty of spotting them. It is possible that increased use of acoustic monitoring and perhaps e-DNA could provide a more complete picture of distribution and occurrence and consequently also assist with the design of future abundance surveys, although the latter is less likely to be useful where density is low. In the Baltic Sea, the Sambah project demonstrated that passive acoustic monitoring can be used to estimate porpoise density and abundance, as well as to map distribution and identify hotspots and preferred habitats and areas with high risks of negative anthropogenic impacts, in a cost-effective way over a large area (SAMBAH 2016). Foote et al. (2012) showed that porpoises could be detected using eDNA within a 10 m distance in a sea pen in a sheltered port area but this is very different to open sea conditions; see Suarez-Bregua et al. (2022) for a review of the utility of the eDNA approach. Another technique based on genetic information, which could provide insights into abundance, is Close Kin Mark Recapture (e.g. Conn et al. 2020, Trenkel et al. 2022).

Demographic characteristics

Current situation

Strandings monitoring provides the main source of demographic data, e.g., age, mortality, reproductive output, health status and potentially indicating changes in distribution and abundance. Both Spain and Portugal have regional strandings networks that have depended heavily on short-term funding and the work of volunteers. They have not always had access to veterinary expertise or sufficient funding to collect and analyse relevant samples. As such, much work has depended on project funding. At the time of writing, national funding has recently been injected into the system in Spain (via *Fundación Biodiversidad*) but coordination across networks still has some way to go.

As seen in section 'Life history', the overall annual mortality rate (including all causes of death) estimated by Read et al. (2013) for the Iberian population from ages of stranded animals during 1990–2010 was 18% (95% CI 5.3%–20.5%). However, the estimated pregnancy rate (54%) from stranded animals, coupled with the estimated age at sexual maturity (ASM) in females, is too low to compensate for 18% annual mortality, and this remains true even if a lower ASM is assumed (see Figure 12). Pregnancy rate is likely to be underestimated from strandings because dead animals inevitably include a higher proportion of animals that were in ill health and hence possibly not able to breed, compared to the living population. The sample sizes here (especially for the estimation of pregnancy rate) were rather small. The validity of the life table-based methodology depends on the stranding data being representative (of deaths) and on population size and age structure being stable, but Read et al. (2013) found evidence that mortality rate varies over time.

Future research and monitoring

The collection of life history, dietary, health and cause of death data from stranded and bycaught animals remains essential to obtain better estimates of demographic parameters and provide a more complete picture of health status and causes of death in this population. They are also essential for genetic studies. One limitation of age-at-death data from strandings is that the youngest animals tend to be underrepresented, possibly because the carcasses are less buoyant and decompose more rapidly. Further work on the population dynamics of Iberian porpoises is needed, for example, using a model-based approach (e.g. Siler 1979) to correct mortality rates (see Saavedra Penas 2017).

As is generally the case in Europe, there is a need for better coordination of the work of strandings networks and sufficient secure funding which covers not only basic monitoring but also full necropsies and the collection and processing of relevant samples. Application of carcass drift models (after Peltier et al. 2012) to the Iberian Peninsula would help to determine the origins of stranded animals and also increase the utility of data collected for estimating bycatch rates (some progress in this direction was made under the recent CetAMBICion project). Most strandings networks depend to a greater or lesser extent on citizen science, which could also be better supported and coordinated. Ongoing initiatives involving ASCOBANS, ACCOBAMS, the IWC and ICES may help to deliver solutions (e.g. ASCOBANS 2023).

Bycatch mortality and other threats

Current situation

The North Atlantic Marine Mammal Commission and the Norwegian Institute of Marine Research (NAMMCO 2019) observed that population size and bycatch data for this population are apparently incompatible – taken at face value the first could not support the second; the population would go extinct in only a few years. Strandings and on-board observations during the last decade suggest total annual bycatch mortalities of between around 200 and 500 animals along the Atlantic coast of the Iberian Peninsula. While estimated bycatch mortality is high, the annual number of *known* bycatch mortalities (i.e. observed bycatch plus bycatch diagnosed in stranded animals) is relatively low (10–15 individuals annually) and thus does not exceed either the ASCOBANS 1.7% threshold (49 animals for this population to achieve 80% of carrying capacity with probability 0.5 (over an indefinite period of time), assuming no biases in abundance or bycatch estimates) or the IMR/NAMMCO Workshop threshold (25 animals for this population to achieve 50% of carrying capacity with probability 0.95 within 100 years). However, it is extremely unlikely that true bycatch mortality is as low as the minimum estimate, and obviously, even the minimum estimate still exceeds the zero limit proposed by OSPAR to achieve 80% of carrying capacity with probability 0.8 within 100 years.

Lack of sufficient reliable data on the amount and distribution of bycatch mortality of porpoises is a key issue. Historically, this reflects the exclusion of the very large number of small (<15 m) fishing vessels from the on-board observer programme and poor implementation of Regulation 812/2004. Although project-based data collection (e.g. from the MARPRO project) filled some gaps, in most cases the proportion of fishing activity monitored was small, and not all fleets were monitored every year, so that the extrapolated bycatch rates were of questionable value. Currently, at a European scale, the increased reliance on fishery observers instead of dedicated observers to collect cetacean bycatch data under Regulation (2019/1241) looks like a backward step. The value of strandings monitoring to provide bycatch data is increasingly recognised and has provided input into recent ICES assessments of cetacean bycatch mortality in the Bay of Biscay (ICES 2020a) but further development of this data source is still needed on the Iberian coast.

Of the other known threats, among the most concerning is the sometimes high PCB concentrations in Iberian porpoises, which are likely to result in at least some reduction in reproductive output. However, as pointed out by Lusseau et al. (2023), porpoises react to underwater noise (to which boat traffic, marine renewable development and the use of pingers on fishing nets, among other sources, all contribute) and their physiological ecology is such that body condition falls rapidly when feeding opportunities are lost. Such effects at individual level can also impact population dynamics. In relation to bycatch mitigation, these authors also warned that noise from pingers can lead to reduced individual reproductive output due to its effects on individual condition and observed that "we do not know how to plan pinger prevalence to minimise bycatch and acoustic impacts" and that there is a need for experimental studies to estimate "dose response relationships" between noise exposure, the resulting behavioural responses and the consequent changes in body condition. In order to achieve this, we also need improved indicators of body condition. Derous et al. (2020) noted that blubber thickness and composition in some parts of the cetacean body does not provide a good measure of health and that more work is needed on the relationships between environmental stressors and blubber biology.

The main measure in place to reduce porpoise bycatch in European waters at present is the use of pingers on fixed nets. To this can be added the recent implementation of 'Move-On' rules and the trialling of pingers and excluder devices for trawl nets in Spanish waters, although these measures are not specifically targeted at porpoises.

In the case of harbour porpoise, many studies, both older and more recent, suggest that pingers can be effective in reducing bycatch mortality in various types of net, increasingly so as pinger design has evolved (e.g. Kraus et al. 1995, 1997, Gearin et al. 1996, Koschinski & Culik 1997, Vingada & Eira 2018, Kindt-Larsen et al. 2019, Brennecke et al. 2022, Königson et al. 2022, Pinn 2023). In Portugal, pingers (F70 and F10) were trialled on several different types of fishing gear within the LIFE+ MarPro project (2011–2017). In purse seines, harbour porpoise mortality was reduced by 75% (based on trials on several vessels over periods of three to 19 months, with a total of 446 hauls with pingers and 551 control hauls); for polyvalent vessels (vessels carrying a range of fixed gears), the reduction was 83% (based on trials with various boats during one to 25 months, with a total of 627 hauls with pingers and 773 control hauls); and for beach seine, the reduction was 55% (based on trials with various vessels over periods of 1–6 months, with a total of 1068 hauls with pingers and 3093 control hauls) (Vingada & Eira 2018).

However, some of the above-mentioned studies also acknowledged negative aspects of pinger deployment: Brennecke et al. (2022) suggested that "pinger use should be limited to critical time periods and regions or that more focus needs to be put on developing acoustic devices which cause less severe behavioral reactions". In general, there is still a need to quantify the effects of disturbance and displacement of animals from their preferred habitat and the consequent effects on body condition and reproductive output (as modelled by Lusseau et al. 2023). This may be most important for small populations and those occupying a confined space (like the Baltic Proper subpopulation) (Sveegaard et al. 2011, Pinn et al. 2021). In Portugal, concern has been expressed that the universal deployment of pingers on beach seines (assuming that they are functioning) may exclude porpoises from important feeding areas.

Perhaps the most glaring issue is the lack of routine bycatch monitoring and mitigation measures (and, where mitigation has been introduced, lack of monitoring of the efficacy of mitigation measures) for many small-scale fisheries in the Iberian Peninsula.

Following the responses by the European Commission and advice from ICES after the 2019 call for Fishery Emergency Measures to reduce bycatch mortality of cetaceans (specifically common dolphins in the Bay of Biscay and harbour porpoise in the Baltic), the Spanish government implemented a strategy to improve knowledge of cetaceans' interactions with fisheries and to reduce bycatch mortality (under Order APA/1200/2020, as detailed in section 'Conservation of porpoises in Europe: legal Protection and its implementation'). The measures included:

- Development of a programme of on-board observation of cetacean bycatch, from September 2020, initially for pair trawl and bottom gillnet vessels using mesh sizes ≥ 80 mm, and more recently extending to (individual) bottom trawls, purse seines and artisanal gears operating in national waters;
- A pilot electronic observation programme for bottom trawl and gillnet vessels;
- Mandatory reporting of incidental catches in the logbook for all fishing vessels;
- Collection of data by stranding networks for collation by the government;
- A pilot study on information collection and harmonisation of protocols for carrying out necropsies.

The European Commission opened an infringement procedure (INFR(2020)4038) against Portugal in November 2023, specifically referring to cetacean bycatch, including that of porpoises[22]. Bycatch in Portugal is being addressed mainly through existing conservation frameworks. Knowledge of the bycatch of protected species in Portugal is based on various previous and ongoing research projects (e.g. SafeSea, MarPro, PescApanha, InovPesca, CetAMBICion), which have provided information on the species and gears involved and bycatch rates, investigated mitigation measures using acoustic devices and developed good practices guides for various fleets. Measures introduced in 2017 and 2019 are described in section 'Conservation of porpoises in Europe: legal Protection and its implementation'.

Under the MSFD programme, several future measures are foreseen, including the establishment of a coordinated strategy (involving Portugal, Spain and France) for the monitoring and assessment of cetacean populations and to address bycatch in the Bay of Biscay and Iberian Coast, recently drafted under the auspices of the CetAMBICion project; formation of a dedicated working group to develop an action plan to reduce bycatch of marine mammals, marine birds and marine turtles; a programme to monitor the use and effectiveness of pingers in the traditional beach seine fishery and to improve control and enforcement; rescue protocols for animals bycaught in traditional beach seines; and an initiative to produce bycatch risk maps and pilot the implementation of spatial and temporal restriction measures for fisheries within the Site of Community Importance (SIC) Maceda-Praia da Vieira (CetAMBICion project partners, Pers. Comm.).

Both Spanish and Portuguese authorities have participated in research projects such as the MSFD 2020 project CetAMBICion, the objectives of which included the determination of abundance and distribution of cetaceans in Spanish waters, the study of the influence of environmental variables and prey availability, calculation of cetacean bycatch mortality and the assessment of the effectiveness of using pingers and other devices for bycatch reduction, and the development of monitoring, evaluation and mitigation measures.

[22] https://ec.europa.eu/commission/presscorner/detail/en/inf_23_5380.

In relation to fishery bycatch and other threats, one of the most significant recent developments was the publication in March 2023 of the OSPAR Quality Status Reports for Indicator Assessments under the MSFD[23]. For the Bycatch Indicator, in relation to bycatch of Iberian porpoise, it is stated that:

> Although it is not possible to quantify this issue using the methods applied to all other AUs given the lack of records through the observer scheme, the evidence suggests a number greater than 200 harbour porpoises are by-caught annually (Vingada and Eira 2018), which, when taken into context with the by-catch threshold of zero, indicates that by-catch in this AU is critically exceeding the agreed threshold.

The estimate from Vingada and Eira (2018) is one of those that appears above in Tables 7 and 10 (Threats to the Iberian harbour porpoise). The reports also include the Pilot Assessment of Status and Trends of Persistent Chemicals in Marine Mammals (Pinzone et al. 2022), and it is expected that a “persistent chemicals indicator” will ultimately be adopted and pave the way for routine monitoring of pollutant levels in cetaceans.

Future research and monitoring

On-board observers and remote electronic monitoring Improved data collection is needed to permit accurate and precise estimates of bycatch rate. As such, ensuring adequate observer coverage of all relevant fleets, including small-scale fleets, preferably using dedicated observers, currently remains a priority. This was demonstrably not been achieved during the last two decades, as evidenced by the extremely patchy observer data presented in section ‘Threats to the Iberian harbour porpoise – Fishery bycatch’. Challenges that must be overcome include logistical difficulties with placing observers on small vessels, covering a sufficiently large proportion of fishing activity generally, and the reliance on voluntary collaboration by the fishing sector. It is possible that provision of addition training of fishery observers could help improve the quality of observer data on porpoise bycatch (Scientific, Technical and Economic Committee for Fisheries (STECF) 2019). Increased use of Remote Electronic Monitoring such as on-board cameras should ultimately provide better bycatch estimates (in terms of both accuracy and precision), provided that privacy and data-protection issues can be addressed, and bycaught species can be reliably identified from the images obtained, e.g., using Artificial Intelligence (Ovalle et al. 2021, 2022). Self-reporting of bycatch by fishers could also be useful and can be assisted by new technology, as in the case of the Clean Catch UJ smartphone application (Ryan et al. 2022)[24]. Of course, making it easier to report bycatch does not necessarily mean the frequency of reporting will increase.

Strandings monitoring The value of strandings monitoring to provide bycatch data is increasingly recognised and has provided input into recent ICES assessments of cetacean bycatch mortality in the Bay of Biscay (e.g. ICES 2020a). There are several ways in which this contribution could be enhanced: better resourcing, including availability of veterinary expertise, adoption of common protocols and training in their use, introducing elements of survey design and quantifying the amount and distribution of survey effort, development of drift models, adoption of a common reporting format and construction of a central database (thus also eliminating duplication of calls for data by different organisations). Routine monitoring of pollutant levels in stranded porpoises would be valuable. Even if PCB concentrations in cetacean blubber are generally declining in Europe, many animals continue to have concentrations that represent a toxicological threat (Williams et al. 2023). As noted above, OSPAR has recently published a pilot study on status and trends of PCBs in cetaceans in Europe and proposed the use of PCB concentrations in marine mammal blubber as an indicator (Pinzone et al. 2022).

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[23] https://oap.ospar.org/en/ospar-assessments/quality-status-reports/qsr-2023/indicator-assessments/

[24] https://www.seafoodinnovation.fund/projects/cetacean%E2%80%91interaction%E2%80%91self%E2%80%91reporting%E2%80%91and%E2%80%91em%E2%80%91verification%E2%80%91sys-tem%E2%80%91rd061/9781032761961_C001.indd

Bycatch mitigation

Globally, a wide range of mitigation measures has been applied to reduce the bycatch of small cetaceans, including acoustic deterrents, gear modifications such as net height, buoy rope modification, weak links and increased rope visibility/reflectivity, acoustic reflectors, time-area closures, visually detectable devices and exclusion devices (e.g. Hamilton & Baker 2019, Kindt-Larsen et al. 2019, Omeyer et al. 2020, Sacchi 2021).

To understand why some mitigation methods work better than others, it is important to understand why cetaceans are entangled and bycaught in the first place, while recognising that the answers may be specific to particular species and particular kinds of gear and may vary over time. Kinze (1994) noted that bycaught porpoises in Denmark tended to include an unexpectedly high proportion of subadults and suggested that individuals may need to learn to avoid entanglement, either by becoming entangled and escaping or through hearing the distress calls of entangled animals. Kastelein et al. (2000a) suggested that echolocating porpoises do not detect gillnets in time to avoid collision. They estimated the target strength of various gillnets and the distance at which the nets would be detected by harbour porpoises and bottlenose dolphins and concluded that a porpoise approaching a net at night at right angles (perpendicular to the net) would detect the nets from a distance of only 3–6 m (depending on the type of net) 90% of the time, whereas a bottlenose dolphin would detect the same nets from 25 m to 55 m away. It has also been suggested that porpoises are distracted due to focusing on the fish trapped in the net (Kastelein et al. 1995) or are aware of the net but consider it not to be a threat (Larsen et al. 2007).

By far the most common approach to date to reduce porpoise bycatch is the use of acoustic deterrents/alarms (pingers) on fixed gear, to date the only bycatch mitigation method for cetaceans that is specifically mandated in certain fisheries in Europe. Pingers actively emit sounds that may both alert porpoises to the presence of a net and have an aversive effect. The first pingers were designed to reduce harbour porpoise mortality in gillnet fisheries (Kraus et al. 1997) and were later developed for use with other cetacean species and other fishing gears (Culik et al. 2001). As previously commented, numerous trials with pingers have suggested that porpoise mortality in fixed nets can be substantially reduced.

In certain areas, e.g., the Inner Danish Sound and, potentially, the Galician rías, porpoises might be constrained to use the area in which nets with the pingers are deployed and thus continue to interact with the nets to feed, despite the sounds emitted by the pingers (Bordino et al. 2002, Forney et al. 2017, Snape et al. 2018, Omeyer et al. 2020). Pingers also may have some adverse effects on porpoises. Displacement could result in stress reactions and reduced food intake, while loud sounds could result in temporary or permanent hearing loss. Captive porpoises avoided active pingers (surfacing farther away from them) but also show an increased respiration rate (Kastelein et al. 2000b). Lusseau et al. (2023) explored the consequences of pingers for a porpoise population using an agent-based model, showing that low rates of pinger deployment (compared to zero deployment) could increase bycatch mortality by leading to movements that increased encounter probability with gillnets not equipped with pingers, while at high rates of pinger deployment, the physiological impacts of acoustic disturbance on reproduction could negatively affect population dynamics. A balance is needed: a relatively low coverage of the fleet may produce the optimum trade-off between positive and negative effects of pinger deployment.

Another solution is increasing the detectability of the nets to echolocation. Metallic-based coatings such as barium sulphate or iron oxide on gillnets have been found to reduce harbour porpoise bycatch (Larsen et al. 2007, Trippel et al. 2008) although this may be due to the increased stiffness of such nets (Larsen et al. 2007, Cox & Read 2004). Gillnets fitted with passive acoustic reflectors (e.g. acrylic glass air-filled "pearls") also reduced porpoise bycatch rates (Gustafsson 2020, Kratzer et al. 2020, 2021). Such nets may be less easy to handle, with more time needed to prepare the nets and to disentangle bycaught animals (Kratzer et al. 2021, Read 2021).

With all such modifications, it is necessary to consider effects on the performance of the gear, e.g., effects on net buoyancy and rigidity, requirements for maintenance and impacts on catches of target species (see Northridge et al. 2013, Mooney et al. 2007, Rowe 2007, Bordino et al. 2013, Knowlton et al. 2016, Kratzer et al. 2020, Read 2021, Sacchi 2021), as well as the potential negative effects of mitigation on the porpoises. As stated by Lusseau et al. (2023), "For marine species that react to noise and are at risk of bycatch, we need to consider the consequences of the interaction of these two stressors more fully".

Alternative approaches include the use of 'move-on' procedures and time/area closures. Time/area closures in the Gulf of Maine proved to be ineffective at reducing bycatch, at least partly due to the (unpredictable) temporal and spatial variation in bycatch rate and partly because fishing effort and the associated bycatch were simply displaced outside the closed area (Murray et al. 2020). Move-on procedures are essentially dynamic closures, and their implementation depends on fishers having access to adequate – and timely – information about the location of cetaceans and bycatch events. The utility of real-time between-vessel data sharing, as used to reduce bycatch of 'choke' species (Marshall et al. 2021) needs to be explored. However, one limitation is the difficulty of observing small species such as harbour porpoise. Possibly porpoise detection could be enhanced by employing acoustic monitoring.

Regardless of the technical solutions, it will be necessary to work with fishers to adapt their fishing practices so that bycatch is reduced (e.g. Ryan et al. 2022). Under the US MMPA, where action on cetacean bycatch is deemed to be required, fishers and scientists are obliged to set up Take Reduction Teams and to devise and follow Take Reduction Plans. Such case-by-case collaborations could be a way forward in Europe.

Part of the problem with implementing these various mitigation methods is that, despite the lethal consequences of bycatch for individual cetaceans and the potentially serious impact at population level, especially for small populations like that of the Iberian porpoise, at the level of individual fishing operations, bycatch of cetaceans is very rare. As such, the costs of mitigation (e.g. installing and maintaining devices, coping with the different handling characteristics of the gear) are arguably much more readily evident than the benefits. This suggests that market incentives (e.g. a premium price for porpoise-safe fish) could make mitigation more acceptable to fishers. Marketing of 'dolphin-safe' tuna, the capture of which did not involve setting nets on dolphins, was part of the solution to reduce bycatch of dolphins in the Eastern Tropical Pacific tuna fishery in the 1980s and 1990s – which is not to undervalue the key roles played by the development of technical measures and training for fishers (National Research Council 1992). Certification of 'dolphin-safe' tuna fishing is currently offered by Friend of the Sea but there is surely scope to develop similar ecolabels for other fisheries.

Finally, in relation to scientific contributions to bycatch reduction, in order to demonstrate a statistically (and practically) significant reduction in bycatch rates, small-scale and short duration trials are often of little value – the performance of the mitigation approach must be monitored over a longer period of time, alongside an equivalent amount of monitored 'control' fishing activity. This in turn implies a need to move away from reliance on short-term project funding for such trials.

Conclusions and recommendations

The Iberian porpoise is an endangered and largely isolated population (and putative subspecies) numbering only a few thousand individuals. Uncertainties notwithstanding, the population evidently suffers high levels of fishery bycatch mortality that may lead to its demise in just a few years. It is essential that we take steps to understand and concurrently facilitate the recovery of this population before it declines to a point at which extinction will become inevitable. If we do not, the case of the vaquita well illustrates the likely outcome (e.g. Rojas-Bracho et al. 2019). Given that the recommended bycatch mortality is zero, the introduction of effective technical measures to reduce bycatch may be insufficient to ensure the long-term survival of the population. However, provided that the measures introduced *are* effective (in terms of both reducing bycatch and avoiding negative consequences like displacement from preferred habitat), slowing the rate of decline would still be a useful first step.

The population occurs mainly along the west coast of the Iberian Peninsula. In terms of its life history and ecology, the most striking distinctive feature of the Iberian porpoise remains its large size, and its diet includes more pelagic fish than is the case for several conspecific populations. Historical records (albeit some of them anecdotal) suggest that the Iberian porpoise was once much more abundant and was also common along the southern coast of Portugal. Today, the population is characterised by a low density and low genetic diversity, with evidence supporting a continuing decline in abundance most likely attributable to fishery bycatch. A comprehensive strategy is needed to save this population, involving effective and well-monitored mitigation actions and supported by increased and well-implemented legal protection (see CMS 2020, ASCOBANS 2021). A joint ASCOBANS-ACCOBAMS conservation plan could be proposed.

In relation to reducing bycatch mortality, the inevitable lack of perfect data cannot be an excuse for inaction. Discarding past data in the hope of collecting more accurate and/or precise data in the future is a fool's errand. Management can be designed to be robust in the face of imperfect data. This rationale underlies the use of Management Strategy Evaluations in fisheries and the use of PBR as a tool under the US MMPA (Wade 1998, Moore et al. 2013). More specifically, management advice can account for imperfect data. However, referring again to fisheries, at least in the European Union, the fact that a particular maximum catch is advised does not necessarily mean that the catch quota which is finally set will follow this advice, nor that once the quota is set, catches will not exceed it. Implementation and compliance/enforcement are not data driven and their success will depend on some combination of (1) the strength (and compatibility) of the relevant legislation related to fisheries (e.g. the Common Fisheries Policy in the EU and the Magnuson–Stevens Fishery Conservation and Management Act (MSFCMA) in the USA) and marine conservation (e.g. the MSFD and MMPA), (2) the willingness of all the stakeholders to work towards achieving the management objectives and (3) public opinion.

Reduction of cetacean bycatch in Europe in general and of porpoise bycatch in the Iberian Peninsula in particular could be facilitated through stronger collaborations with stakeholders (e.g. through Take Reduction Teams) and harnessing public opinion via porpoise-safe ecolabelling.

The European Commission, national governments, international organisations, NGOs, the fishing industry, scientists and (at least to some extent) the general public all realise that action on cetacean bycatch mortality is needed – and that Iberian harbour porpoise is one of the populations for which it is needed most urgently, preferably allowing the recommended bycatch mortality limit of zero to be achieved. A separate assessment of the (Red List) status of this population by the IUCN cetacean group would be welcome.

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