Agenda Item 8 Relevant EU Policy matters

Document Inf.8.a **Background document on relevant EU policy matters**

Action Requested

- Take note

Submitted by DG Environment

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Secretariat's Note

The Rules of Procedure adopted at the ASCOBANS 8th Meeting of Parties remain in force until and unless an amendment is called for and adopted.

Update on the implementation of the EU nature, marine and fisheries policies relevant for ASCOBANS activities

Prepared by the European Commission's Directorate General for Environment and Directorate General for Maritime Affairs and Fisheries

1. State of play with the implementation of the Habitats Directive and the Action plan for nature, people and the economy (relevant issues)

Following a thorough evaluation of the **Birds and Habitats**[1] **Directives**, the European Commission has adopted the **Action plan for nature, people and the economy**[2] (AP) to improve their implementation and boost their contribution towards reaching the EU's biodiversity targets for 2020. The AP focuses on four priority areas and comprises 15 actions and over 100 measures to be rolled out until 2019, many of those being relevant for cetacean conservation.

1.1. Strict protection of species

The Habitats Directive affords **strict protection** to all cetaceans (Cetacea are listed in Annex IV). Member States have to take the necessary measures to establish a system of strict protection in their entire natural range, prohibiting their deliberate capture or killing, disturbance and deterioration or destruction of their breeding sites or resting places. They should also establish a system to monitor the incidental capture or killing and take the necessary measures to ensure that it does not have a significant negative impact on the species concerned.

The Commission is currently updating **the guidance on species protection rules under the Habitats Directive** as foreseen in the AP. A first draft of the updated guidance document will be available for discussion in October 2018 with a view to having a final draft ready by the end of 2018 and a formal adoption by the Commission in early 2019. Comments on the relevant parts of the guidance document are welcome and can be provided through the Marine Expert Group once a draft will be distributed for comments.

1.2. Establishment and management of Natura 2000 sites

More than **3150 marine Natura 2000 sites** currently cover **532 417 km^2** or **9.2%** of EU seas[3]. This largest coordinated network of protected areas in the world contributes most significantly to the marine protected areas' coverage in Europe and to reaching the international targets.

For two cetaceans listed in Annex II of the Habitats Directive, **the bottlenose dolphin** *Tursiops truncatus* and **the harbour porpoise** *Phocoena phocoena*, special areas of conservation (Natura 2000 sites) need to be designated and managed according to the ecological needs of these species. This means that appropriate **conservation measures** should be established and implemented to reach the site-specific **conservation objectives**. The conservation measures in Natura 2000 sites, together with measures taken under strict protection regime in the entire natural range, should aim to achieve or maintain the favourable conservation status of these species.

[1] Council Directive 92/43/EEC of 21. May 1992 on the conservation of natural habitats and of wild fauna and flora (OJ. L 206, 22. July 1992, p. 2).
[2] http://ec.europa.eu/environment/nature/legislation/fitness_check/action_plan/index_en.htm
[3] https://www.eea.europa.eu/data-and-maps/dashboards/natura-2000-barometer

In the area of the Baltic and Northern Seas and the Atlantic, there are 83 Natura 2000 sites with the presence of the bottlenose dolphin (all in the Atlantic) and 234 sites with the presence of the harbour porpoise (169 in the Atlantic and 65 in the Baltic). Although the designation of sites for these two species has advanced in recent years, there are still some gaps, mainly offshore. There are multiple challenges with identifying Natura 2000 sites for such mobile and wide-ranging species as Member States should propose sites where there exists a "*clearly identifiable area representing the physical and biological factors essential to their life and reproduction*". These species are usually difficult to observe, data concerning their distribution patterns at sea are sparse and research is expensive and needs to be conducted over long time periods. Significant efforts were made in the past to collect the data on distribution and population sizes with many projects co-financed by EU funds (for example SCANS II or SAMBAH). **The completion of the Natura 2000 network**, especially its marine part and including the establishment of the **appropriate conservation measures**, is one of the main actions of the AP. Concerning the implementation of conservation measures and site management plans, for these two species the progress is not entirely satisfactory. The efforts should be increased to establish and implement the necessary conservation measures so that the sites can contribute to reaching the favourable conservation status of these species.
The Commission is taking opportunity of the bilateral dialogues with Member States under the AP to highlight these obligations and is also taking legal action where necessary.

1.3. Reporting

Article 11 of the Habitats Directive requires Member States to monitor the habitats and species listed in the annexes and Article 17 requires a report to be sent to the European Commission every 6 years following an agreed format. The core of this report is the assessment of the **conservation status** of each habitat type and species and the data underpinning such assessment. The next national reports using an updated report format[4] are due in April and July 2019 covering the period 2013-2018. In order to report on the conservation status, it is important to set the **favourable reference values** (FRVs). The concept of FRVs is derived from definitions in the Directive, particularly the definition of the favourable conservation status that relates- for species, to the 'long-term distribution and abundance' of the populations of species 'in their natural range'. The Commission has commissioned a study looking at the definition and application of the concept of FRVs with the aim to facilitate their setting and to improve – in certain cases - cooperation in setting FRVs at a biogeographical level. While the final report is not yet available, the draft examples of FRVs for few cetacean species are shown in Annex 1.

Synergies with reporting under the MSFD are mentioned under 2.2.

1.4. Supporting the cooperation in Natura 2000 management

The implementation of marine Natura 2000 is supported by the Marine Expert Group (MEG) set up by the Commission to promote the exchange of experience, information and best practices in site designation and management, including addressing pressures from fisheries and other activities, and to promote synergies with the Marine Strategy Framework Directive (MSFD).
Natura 2000 seminars under the biogeographical process and follow-up activities are another way of fostering cooperation between Member States on the management of marine sites. The

[4] https://bd.eionet.europa.eu/activities/Reporting/Article_17/reference_portal

first marine biogeographical seminar was held in St. Malo in France in May 2015[5]. **The second marine Natura 2000 seminar** will take place in Palma de Mallorca, Spain, **13-15 November 2018**. It will focus on setting the conservation objectives and measures, including the FRVs for selected habitats and species, among them for two cetaceans that require site designation. Further information on the management of marine Natura 2000 sites can be found on the dedicated webpage[6] and the marine page of the Natura 2000 communication platform[7].

1.5. Financing

The role of EU co-financing is significant for cetacean conservation. The EU has supported projects to collect data on their distribution, map the main threats and resolve conflicts. The most important EU funds used for marine conservation are the **LIFE programme**[8] and the **European Maritime and Fisheries Fund**.

The Commission has recently adopted a proposal for the **new multiannual financing framework** for the period 2012-2027[9]. The LIFE Climate and Environment budget is proposed to be **increased** significantly to €5.45 billion[10]. The new LIFE programme will support projects that promote best practices in relation to nature and biodiversity, as well as new, dedicated 'strategic nature projects' for all Member States to help mainstream nature and biodiversity policy objectives into other policies and financing programmes, ensuring a more coherent approach across sectors. This would require a robust strategic planning of investments in nature through **prioritised action frameworks (PAFs)**. PAFs are strategic multiannual planning tools, aimed at providing a comprehensive overview of the measures that are needed to implement the Natura 2000 network and the nature directives, specifying the financing needs for the necessary measures and linking them to the corresponding EU funding programmes. PAFs shall focus on the identification of those financing needs and priorities that are directly linked to the specific conservation measures established for Natura 2000 sites, in view of achieving the site-level conservation objectives for those species and habitat types for which the sites have been designated. The AP commits to ensure that Member States provide more reliable and harmonised estimates of their financing needs for Natura 2000. The Commission has signed a new contract on "Strengthening investments in Natura 2000 and improving synergies with EU funding instruments" which will assist Member States in understanding and making the most out of the new funding programmes as regards Natura 2000. Workshops will be organised in Member States to facilitate the drafting of PAFs and this will allow appropriate stakeholder input.

2. State of play with the implementation of the Marine Strategy Framework Directive (relevant issues)

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[5] http://ec.europa.eu/environment/nature/natura2000/platform/events/157_first_marine_biogeographical_process_seminar_en.htm

[6] http://ec.europa.eu/environment/nature/natura2000/marine/index_en.htm

[7] http://ec.europa.eu/environment/nature/natura2000/platform/knowledge_base/212_marine_regions_en.htm

[8] See the latest brochure for details on the LIFE and the marine environment http://ec.europa.eu/environment/life/publications/lifepublications/lifefocus/documents/marine_environment_web_2018.pdf

[9] https://ec.europa.eu/commission/publications/factsheets-long-term-budget-proposals_en

[10] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM%3A2018%3A385%3AFIN

2.1. Determination and achievement of the good environmental status

The **Marine Strategy Framework Directive**[11] (MSFD; Directive 2008/56/EC) aims to achieve the good environmental status of the EU's marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend. The seas in GES are clean, healthy and productive. The Directive enshrines in a legislative framework the ecosystem approach to the management of human activities having an impact on the marine environment, integrating the concepts of environmental protection and sustainable use. According to the Directive, each Member State must implement a marine strategy for its marine waters, in cooperation with other Member States sharing the same marine region, reviewed every 6 years. Those strategies include 5 steps:

1. an initial assessment of their marine waters,
2. the determination of the good environmental status of their marine waters,
3. the setting of environmental targets,
4. the establishment and implementation of coordinated monitoring programmes, and
5. the identification of measures or actions that need to be taken in order to achieve or maintain good environmental status.

Within the context of the MSFD, Member States sharing a marine region or sub-region are also encouraged to cooperate to deliver on the objectives of the Directive. The Commission, through DG Environment, is ensuring that Member States continue in their collaborative efforts to implement the MSFD, including through the work of regional sea conventions and through the common implementation strategy. The Commission is also committed to the implementation of the MSFD through funding opportunities as appropriate and with the collaboration of all Member States.

Following the first (2014) assessment of Member State's reports on the state of their marine waters, determination of Good Environmental Status and associated targets, the Commission concluded to the need **to improve the implementation of the MSFD**. In particular, more efforts are urgently needed to reach the GES by 2020 and coherence with other EU legislation and regional approaches should be strengthened. In this regard, the new **Commission Decision (EU) 2017 /848**[12] adopted in May 2017 should help ensuring an effective implementation of the directive for the next cycle.

With the new GES decision, we have now entered the second implementation cycle and Member States are presently updating the information about steps 1-3 above. The Commission has just released its **assessment of Member States' programmes of measures** for the first cycle[13], checking for example whether the right pressures are being tackled in the region and whether these measures are consistent. This enables us to have a better understanding of our marine waters. The outcomes of this first assessment indicate that MS have made considerable efforts to develop their programmes of measures, by integrating different national, EU and international policies and processes for the purpose of protecting the environment. They have also established new measures to specifically target pressures on the marine environment, thus

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[11] http://ec.europa.eu/environment/marine/eu-coast-and-marine-policy/marine-strategy-framework-directive/index_en.htm

[12] Commission Decision (EU) 2017/848 of 17 May 2017 laying down criteria and methodological standards on good environmental status of marine waters and specifications and standardised methods for monitoring and assessment, and repealing Decision 2010/477/EU.

[13] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM:2018:562:FIN&qid=1533034580736. The report is accompanied by a Staff Working Document (https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52018SC0393) with detailed guidance per Member State.

showing the added value of the MSFD. However, not all pressures on the marine environment are covered properly and there is a **lack of regional or EU coordination** which leads to a fragmented approach for certain pressures of transboundary nature.
Last but not least, Member States still have to implement fully their monitoring programmes to be efficient and operational as soon as possible. Global efforts are necessary in particular for the descriptor on non-indigenous species, marine litter and underwater noise.

2.2. Marine protected areas and links with the Habitats Directive

Achieving the GES requires the implementation of programmes of measures (following Article 13 of the MSFD) to manage the pressures arising from human activities, in order to reduce their impacts on the marine environment. Those programmes shall include **spatial protection measures**, contributing to **coherent and representative networks of marine protected areas (MPAs)**, adequately covering the diversity of the constituent ecosystems. Most Member States have already a well-developed network of MPAs, largely linked to the Habitats and Birds Directives. Still, more efforts are needed to design ecologically coherent networks that are effectively managed.

The EU has been a strong advocate for delivery of the global MPA targets and has cooperated closely with relevant regional and global organisations. The EU has also taken action and supported initiatives to fulfil international commitments and targets such as the "Aichi Target 11" of the Convention on the biological diversity (*i.e.* to protect 10% of marine and coastal areas through effectively managed MPAs and other area-based conservation measures which form ecologically coherent networks by 2020) and the UN Sustainable Development Goal 14.5 (*i.e.* to conserve at least 10% of coastal and marine areas by 2020, consistent with national and international law and based on the best available scientific information). The overall target of conserving 10% of marine and coastal areas (in terms of area covered through the designation of MPAs) **has been already attained in European waters**, although with large differences among regions[14].

In March 2018, the European Commission, Member States and regional sea conventions met for a **joint meeting on biodiversity assessment and reporting** under the MSFD and HBD[15]. The three directives have the same aim of achieving long-term favourable conditions for keeping the species and habitats in a good conservation status. All involved actors agreed that aligning the implementation processes of these policies (*i.e.* monitoring and assessments) will increase their efficiency and coherence. During the meeting, the main challenges and possible ways forward were identified and illustrated with some Member States' examples. The Commission is now exploring further steps.

2.3. Descriptor 11 on underwater noise

Determination of the GES by Member States is based on 11 descriptors, one of which is dedicated to the introduction of energy, including **underwater noise**, which should be at levels that do not adversely affect the marine environment. This is further specified in the Commission Decision 2017/848, which defines two types of criteria for Descriptor 11: (a)

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[14] https://icm.eionet.europa.eu/ETC_Reports/SpatialAnalysisOfMarineProtectedAreaNetworksInEuropesSeas_VolumeA_2017

[15] All documents and presentations are accessible in CIRCABC: https://circabc.europa.eu/w/browse/fb68e18d-0b1a-46fb-bd11-b1aec6fe0978

anthropogenic impulsive sound in water and (b) anthropogenic continuous low frequency sound in water. The primary criteria for both types are that the spatial distribution, temporal extent, and the levels of anthropogenic impulsive sound or continuous low-frequency sound sources do not affect populations of marine animals. In both cases the Member States shall establish **threshold values** for these levels through cooperation at Union level, taking into account regional and sub-regional specificities. Methodological standards as well as specifications and standardised methods for monitoring and assessment are given in detail for both types of sound sources.

In the framework of the common implementation strategy adopted for the MSFD, a **Technical Group on underwater noise** (TG Noise) advises Member States on these issues. In particular, until 2015 its work has been focused on monitoring, with regard to setting up a register of loud impulsive noise and the development of a joint monitoring programme for continuous noise. Significant progress has been made in this field during the 1st cycle of implementation of the marine directive. For future assessment and target setting, in view of reaching good environmental status, implementation of the current indicators on impulsive and ambient noise remain essential. Meanwhile, TG noise focuses now on assessment of impacts of noise and development of thresholds in relation with the indicators developed in the framework of MSFD.

3. State of play with the implementation of the common fisheries policy (CFP) and integrated maritime policy (relevant issues)

The EU's **common fisheries policy (CFP)** aims to ensure that fishing and aquaculture are **environmentally, economically and socially sustainable**. This also means ensuring that the impact of fishing on protected species and habitats is minimised and in line with the obligations set under the EU's environmental legislation. There are important links in particular between the Birds and Habitats Directives, the MSFD and the CFP tools related to data collection, fisheries management and financing. These links **have recently been strengthened** and their success will depend on the effective implementation of existing and new rules. Below is the outline of the main relevant developments under the CFP.

3.1. Data collection

The EU fisheries management relies on the data collected and supplied by Member States under the **data collection framework** (DCF). The newly revised DCF[16] that came into force in 2017 establishes rules on the collection, management and use of biological, environmental, technical and socio-economic data concerning the fisheries sector, contributing to the objectives of the CFP. **It has new requirements on collection of data to assess the impact of EU fisheries on marine ecosystems in Union waters and outside Union waters, and in particular on incidental bycatch of birds, mammals, reptiles and fish protected under Union legislation and international agreements.** Such new requirements aim to fill the existing data gaps and **facilitate compliance** with the provisions of Article 12 of the Habitats Directive. Currently, the data are collected under the **multiannual Union programme** for the collection, management and use of data in the fisheries and aquaculture sectors for the period 2017-2019[17].

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[16] New DCF: http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32017R1004
and EU-MAP: http://eur-lex.europa.eu/legal-content/EN/ALL/?uri=uriserv:OJ.L_.2016.207.01.0113.01.ENG

[17] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=uriserv:OJ.L_.2016.207.01.0113.01.ENG

Data collection methods and quality need to be appropriate for the intended purposes follow the best practices and relevant methodologies advised by the relevant scientific bodies.
For all types of fisheries and vessels, incidental bycatch of all birds, mammals and reptiles and fish protected under Union legislation and international agreements, including absence in the catch, needs to be collected during scientific observer trips on fishing ships or by the fishers themselves through logbooks. Where data collected during observer trips are not considered to provide sufficient data on incidental bycatch for end-user needs, other methodologies need to be implemented by Member States. The selection of these methodologies shall be coordinated at marine region level and be based on end-user needs. The **Regional Coordination Groups** (RCGs) are established under Article 9 of Regulation (EU) 2017/1004 per each marine region to facilitate a regional cooperation in the collection of relevant data. RCGs are composed of Member States of each marine region and the Commission and in 2017 RCGs met for the first time, replacing previous regional coordination meetings. All outcomes of the RCGs and the liaison meeting, that follows them and coordinates regional work on a horizontal level, can be found online[18] in order to ensure transparency. The RCGs have to draft and agree on their rules of procedures that are meant to define how they work, how decisions are taken and who can be invited. Once adopted, all RCG rules of procedures will be published on the DCF website. Interested parties can contact the chair of the relevant RCG. More details on 2018-2019 meetings schedule and RCG chair contacts can be provided upon request.

Annex IIA in the memorandum of understanding between the EU and the International Council for the Exploration of the Sea (ICES) requests ICES, under "Fisheries-based advisory deliverables", to: *"provide any new information regarding the impact of fisheries on other components of the ecosystem including small cetaceans and other marine mammals, seabirds and habitats. This should include any new information on the location of habitats sensitive to particular fishing activities".* On the basis of the work undertaken by the ICES Working Group on Bycatch of Protected Species (WGBYC)[19], ICES issues advice on an annual basis.
In 2018, ICES reported in its advice[20] that only the bycatch risk to harbour porpoises (*Phocoena phocoena*) and common dolphins (*Delphinus delphis*) in the southern part of the Celtic Seas and to common dolphins in the Bay of Biscay was evaluated, finding that these may exceed internationally adopted thresholds of acceptability. ICES has advised on other areas in previous years. Some other major fishing countries failed to provide any information. ICES evaluation and external assessments of the numbers of bycaught dolphins recorded on the shores of the Bay of Biscay indicate that a dedicated bycatch observer programme and bycatch mitigation is required for relevant fisheries in this area. Mitigation is required under Regulation 812/2004 in some fisheries in the southern Celtic Seas and this mitigation may not be adequate. The impact of fisheries on seabirds and other vertebrates have not been evaluated due to insufficient available information.
ICES notes that the EU multiannual programme (MAP) for data collection aims to improve consistency of bycatch data at a regional scale and should improve the ability of ICES to advise on the impact of fisheries. ICES is moving away from using Member State reports under Council Regulation (EC) No. 812/2004 as the primary source of data on bycatch of cetaceans and other animals. In future, data will be provided through the ICES regional database and estimating system (RDBES) as a result of the implementation of the EU MAP.

[18] https://datacollection.jrc.ec.europa.eu/docs/rcm
[19] The report from the latest meeting is available here: http://ices.dk/sites/pub/Publication%20Reports/Expert%20Group%20Report/acom/2018/WGBYC/wgbyc_2018.pdf
[20] http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2018/2018/byc.eu.pdf

3.2. Management tools

Multiannual plans

Multi-annual plans (MAPs) are one of the main instruments to achieve CFP objectives. After the Baltic MAP and the North Sea MAP entered into force[21], the Commission proposed two new MAPs: one for demersals in the Western Mediterranean[22] and one for demersals in Western Waters[23]. The work also continues on the Adriatic MAP proposed last year[24]. MAPs include the target of fishing at MSY and the deadline, measures for the implementation of the landing obligation, safeguards for remedial action and may also include technical measures.

Measures to comply with the obligations under the environmental legislation

The 2013 reform of the CFP introduced regionalisation whereby Member States concerned may submit **joint recommendations** for the adoption of Commission delegated acts. The new generation of MAPs also contains provisions on regionalisation, which allow Member States and stakeholders to work together on tailor-made management measures that suit their sea basins. Regionalisation can also apply when it is necessary to fulfil the obligations under the environmental legislation, *i.e.* to establish conservation measures in Natura 2000 sites or to comply with the obligations under the MSFD.

Following the consultation of Member States and stakeholders, the Commission has recently adopted a **Staff working document on the establishment of conservation measures under the common fisheries policy for Natura 2000 sites and for Marine Strategy Framework Directive purposes**[25]. This guidance document will be of assistance for the establishment of fishery management measures under Article 11 of the CFP Regulation in order to comply with environmental legislation, where Member States need to significantly increase their efforts.

A review of commonly used approaches for managing fisheries in marine Natura 2000 sites, with some illustrative examples, was prepared in 2018 under the Marine Expert Group[26].

Technical measures

Technical measures are a broad set of rules which govern how, where and when fishermen may fish. The numerous regulations on the technical measures in the EU needed to be modernized in light of the reformed common fisheries policy and therefore the European Commission has put forward **a new framework proposal** for the conservation of fishery resources and the protection of marine ecosystems through technical measures[27]. It contains targets and obligations to implement measures to minimise and where possible eliminate the incidental bycatch of species protected under the Birds and Habitats Directives, along with some baseline measures for certain sea basins, mostly those measures contained in the current technical measures regulations. The proposed regulation will complete the framework for the implementation of horizontal conservation measures required under Article 12 of the Habitats

[21] Respectively Regulation (EU) 2016/ of 6 July 2016 (OJ L 191, 15.7.2016, p. 1) and Regulation 2018/973 of 4 July 2018

[22] COM/2018/0115 final - 2018/050 (COD)

[23] COM/2018/0149 final - 2018/074 (COD)

[24] COM/2017/097 final - 2017/043 (COD)

[25] http://ec.europa.eu/environment/nature/natura2000/marine/docs/Marine%20SWD%20288%20final.pdf, available in all relevant languages here http://ec.europa.eu/environment/nature/natura2000/marine/index_en.htm

[26] http://ec.europa.eu/environment/nature/natura2000/marine/docs/Review%20of%20fisheries%20management%20measures%20in%20Natura%202000%20sites.pdf

[27] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM:2016:134:FIN

Directive, therefore also outside Natura 2000 sites and other MPAs. This proposal is currently discussed by the co-legislators.

One of the objectives with the new technical measures proposal was to provide a simpler and single legislative framework for technical measures regulations that are currently spread over 30 individual legislative acts. In this regard, the proposal also incorporates the main mitigation and monitoring requirements contained in Council Regulation (EC) 812/2004. In addition, the Commission has also proposed a geographic extension of the mandatory use of acoustic deterrent devices to all sea basins, removing the partial coverage in the Baltic Sea and extends their use in the South Western Waters (ICES sub-areas VIII and IXa) and into the West of Scotland (ICES sub-area VIa).

3.3. Fisheries control

A recent Commission REFIT evaluation, a special report of the European Court of Auditors and a resolution by the European Parliament have all shown that the fisheries control system dating back to 2009 had its deficiencies and was overall not fit for purpose. The Commission's proposal to **revise the fisheries control system** was adopted on 30 May 2018. The Commission decided to propose a number of changes to the Control Regulation, as well as targeted amendments to the Regulation on illegal, unregulated and unreported fishing and to the EFCA founding Regulation. The overall objective of the revision is to **modernise, strengthen and simplify** the EU fisheries control system and to increase the level playing field in fisheries controls.

The proposed rules on **fishing restricted areas** are more exhaustive as they apply to any marine area where fishing activities are temporary or permanently restricted or prohibited. In line with the EU plastics strategy, the Commission is proposing to reinforce rules on lost fishing gears, and to make the reporting of lost gears easier and systematic. In addition, under the proposed rules, fishing in fishing restricted areas is considered a serious infringement.

3.4. Financing (European Maritime and Fisheries Fund, EMFF)

Within the AP, the Commission undertook to facilitate **full and effective use** of the financial resources allocated for biodiversity protection and Natura 2000 under the EMFF for the period 2014-2020. The Commission has assessed the current use and intends to facilitate better spending through expert group meetings or bilateral dialogues with the managing authorities.

Commission's proposal for the next EMFF was recently adopted[28]. The EMFF will continue to support the European fisheries sector towards more sustainable fishing practices, with a particular focus on supporting small-scale fishers. The main relevant elements include continuation of spending for biodiversity protection, species protection under the Birds and Habitats Directives and the management, restoration and monitoring of Natura 2000 sites, support for actions to achieve or maintain a good environmental status under the MSFD and support for the collection by fishers of lost fishing gears and marine litter from the sea. The proposal also foresees support for the **collection, management and use of data** on the state of the marine environment, with a view to fulfilling monitoring and site designation and management requirements under the Birds and Habitats Directives, and to supporting maritime spatial planning.

Other relevant information

[28] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=COM%3A2018%3A390%3AFIN

The Commission (DG Maritime Affairs and Fisheries) will launch in late 2018/early 2019 a study on "Environmental pressures and maritime spatial planning". The aim of this study is twofold. First, it will aim at providing information to Member States on how to incorporate the ecosystem-based approach in their maritime spatial planning (MSP) processes, and second, it will explore how the implementation of MSP is linked to the GES and the related objectives and measures of the MSFD. It is expected that the study will be completed by end 2019.

The EU funded project "Towards a transatlantic partnership of marine protected areas" was finalised and its outputs are available on http://transatlanticmpanetwork.eu/. This 2-year project gathered managers from Africa, the Americas and Europe to discuss governance, financial sustainability and management practices of marine protected areas and their networks. One of the networking groups focused on the exchange and sharing of best practice to improve the effective management of MPAs protecting whales in the Atlantic.

New publication on LIFE and the marine environment is available here: http://ec.europa.eu/environment/life/publications/lifepublications/lifefocus/documents/marine_environment_web_2018.pdf

For updated information on all our activities, please visit our web pages:

http://ec.europa.eu/environment/

https://ec.europa.eu/fisheries/

Defining and applying the concept of Favourable Reference Values for species and habitats under the EU Birds and Habitats Directives

Examples of setting favourable reference values

Service contract No. 07.0202/2015/715107/SER/ENV.B.3 financed by the European Commission – contractor: Alterra, institute within the legal entity Stichting DLO (now: Wageningen Environmental Research)

R.J. Bijlsma[1], E. Agrillo[2], F. Attorre[2], L. Boitani[3], A. Brunner[4], P. Evans[5], R. Foppen[6], S. Gubbay[7], J.A.M. Janssen[1], A. van Kleunen[6], W. Langhout[4], M. Pacifici[3], I. Ramírez[8], C. Rondinini[3], M. van Roomen[6], H. Siepel[9], C.A.M. van Swaaij[10] & H.V. Winter[11]

1 Wageningen Environmental Research
2 Comunità Ambiente
3 Istituto Ecologia Applicata
4 Stichting BirdLife Europe
5 Sea Watch Foundation
6 Sovon Dutch Centre for Field Ornithology
7 Susan Gubbay
8 BirdLife International
9 Radboud University Nijmegen
10 Dutch Butterfly Conservation
11 Wageningen Marine Research

Wageningen Environmental Research
Wageningen, July 2018

Contents

Preface

The Birds and Habitats Directive ensure the conservation of a wide range of rare, threatened or endemic animal and plant species as well as characteristic habitat types in Europe. In reporting both directives use 'distance to target' measures regarding conservation status. The Habitats Directive considers explicit favourable reference values while the Birds Directive requires to maintain bird populations at a level which corresponds to their ecological, scientific and cultural requirements.

This report presents examples of setting favourable reference values following the methodology of the accompanying Technical report and in agreement with the Explanatory Notes and Guidelines for reporting under Article 17 of the Habitats Directive for the period 2013–2018 (http://cdr.eionet.europa.eu/help/habitats_art17).

The study was commissioned by the EC under the service contract 'Defining and applying the concept of favourable reference values for species and habitats under the EU Birds and Habitats Directives' (Service contract No. 07.0202/2015/715107/SER/ENV.B.3).

The work was followed and reviewed by the Ad hoc group on 'favourable reference values' of the Expert Group on Reporting under the Nature Directives, and supported the general objective of the ad hoc group (chaired by EEA), namely to improve the guidance related to the setting and reporting of favourable reference values under the nature reporting, and contribute to further harmonise approaches between Member States.

1 Cetaceans

Peter G.H. Evans

1.1 Common bottlenose dolphin (*Tursiops truncatus*) in the European Atlantic

Step 1.1 Biology of the species

The common bottlenose dolphin (*Tursiops truncatus*) is one of the most widely distributed members of the family Delphinidae, which comprises 38 species of dolphins including larger species such as long-finned pilot whale and killer whale. Bottlenose dolphins are social animals, forming groups that at times may number in the tens or hundreds, although normally they live in smaller sub-groups, often referred to as a fission-fusion society with individuals associating with one another for varying lengths of time. Individuals commonly cooperate with one another in activities including babysitting and coordinated hunting. Adult females and young preferentially associate with other female-young pairs whilst in some regions adult males have been reported forming stable alliances with other males. Males reach sexual maturity at around 10-15 years of age, and females rather earlier, at 5-13 years. Age at sexual maturity can vary between regions and within a region, between individuals. Males may reach >20 years before attaining actual breeding status. On reaching sexual maturity, females typically give birth to a single calf every three years although this too can vary from 2-8 years even within the same population. Males may live to 40-45 years and females to c. 50 years. Generation length has been estimated at 20.6 years (21.1 years in populations at a stable state). Bottlenose dolphins have catholic diets including pelagic, demersal or benthic fish, although tending to reflect the habitat in which groups are living. More detailed accounts of the biology of the species can be found in Wells *et al.* (1987), Wells & Scott (1989), Connor *et al.* (2000), Reynolds *et al.* (2000), and Wells & Scott (2009).

Step 1.2 Spatial scale of functioning

The common bottlenose dolphin has a worldwide distribution in tropical and temperate seas of both hemispheres. In the North Atlantic, it occurs from Nova Scotia in the west and the Faroe Islands in the east (occasionally as far north as northern Norway and Svalbard), southwards to the Equator. The species also occurs throughout the Mediterranean and in the Black Sea. It is largely absent from the eastern North Sea, Inner Danish Waters and the Baltic (Reid *et al*. 2003).

In many parts of the world, a coastal bottlenose dolphin ecotype has been distinguished from an offshore one, each having different ecologies, food preferences and movement patterns (Walker 1981, Mead & Potter 1995, Curry & Smith 1998, Hoelzel *et al*. 1998, Perrin *et al*. 2011). Within these ecotypes there may be further population sub-structuring leading to separate stocks or management units. The definition of a management unit is a demographically independent biological population (Palsbøll *et al*. 2007, Evans & Teilmann 2009, NOAA 2016). It is usually based upon a variety of lines of evidence including genetics, morphometrics, distribution and movements (i.e. connectivity), and life history parameters.

While high mobility of the species facilitates interaction and gene flow over large distances (Hoelzel 1998, Querouil *et al.* 2007), bottlenose dolphins can also display fine-scale genetic population structure resulting from localised adaptations over small spatial scales (Ansmann *et al.* 2012). Genetic differentiation between neighbouring populations regularly occurs and may be related to habitat borders (Natoli *et al.* 2005, Bilgmann *et al.* 2007, Wiszniewski *et al.*, 2009), sex-biased linked dispersal (Möller *et al.*, 2004; Bilgmann *et al.*, 2007; Wiszniewski *et al.* 2010), niche specialisation (Louis *et al*. 2014a), anthropogenic activities (Chilvers & Corkeron 2001), and through isolation by distance without apparent boundaries separating populations (Krützen *et al.* 2004, Rosel *et al.* 2009). Consequently, bottlenose dolphins tend to be subdivided into small discrete coastal populations residing relatively close to shore and a much larger wide-ranging offshore population. The relationships both within and between those coastal and offshore populations often remain unclear (Rosel *et al.* 2009, Toth *et al.* 2012, Richards *et al.* 2013, Louis *et al*. 2014b).

In the Northwest Atlantic, sixteen separate coastal stocks or management units and one offshore one have been recognized (Waring *et al*. 2016). In Atlantic European waters (MATL biogeographic region, reported under HD Art 17 by ES, FR, IE, PT and UK), bottlenose dolphins consist of a large and wide-ranging offshore population and much smaller coastal populations The preferential habitat for the offshore ecotype is over the outer continental shelf and shelf break (Reid *et al.* 2003, Certain *et al*. 2008). The coastal populations are fairly resident and tend to inhabit smaller areas close to shore. Ten coastal assessment (= management) units for bottlenose dolphins have been identified, largely from photo-ID studies (ICES 2013, 2016). These coastal populations are also potentially exposed to a greater level of human activity due to their proximity to humans and the fact that they live in relatively small areas. There has been no attempt as yet to examine population structure for bottlenose dolphins in the Mediterranean Sea, beyond genetic evidence for a general split between the eastern and western Mediterranean, and with the Black Sea (Natoli *et al*. 2004). Figure 1.1.1 depicts the different assessment units currently proposed for European Atlantic populations. With more information, these will likely be modified. Population estimates for the small coastal populations have been derived mainly using capture-mark-recapture analytical techniques, where animals are "captured" photographically, and identified individually by unique markings mainly on the dorsal fin, such as patterns of nicks. The same or different individuals may then be "re-captured" photographically in successive encounters.

In northern Europe, the continental shelf edge is generally some distance from the coast thus separating coastal and offshore ecotypes spatially, although groups of the latter may on occasions enter coastal waters. Further south in the Bay of Biscay and around the Iberian Peninsula, the shelf edge comes close to the coast and it becomes more difficult to separate the two ecotypes unless individuals are recognized photographically. Population estimates in this region are mainly derived from line-transect surveys to determine absolute abundance rather than by photo-ID, and no attempt therefore has been made to try to distinguish between ecotypes or management units. Those population estimates are highlighted in red or orange in Figure 1.1.1.

In coastal waters, bottlenose dolphins often favour river estuaries, semi-enclosed bays, headlands and sandbanks where there is uneven bottom relief and/or strong tidal currents (Lewis & Evans 1993, Liret *et al*. 1994, Wilson *et al*. 1997, Liret 2001, Ingram & Rogan 2002).

Genetic studies using mitochondrial DNA indicated some population sub-structuring between animals from east and west Scotland and from Wales, although for the latter two regions, sample sizes were low; there was also a low level of genetic diversity in the east Scottish sample (Parsons *et al*. 2002). Using both mtDNA and microsatellite techniques, fine-scale population structure was revealed among three distinct populations in Ireland – one in the Shannon Estuary, another from the Connemara-Mayo region and a third from strandings of unknown origin but thought to be part of a large offshore population (Mirimin *et al*. 2011). This study found moderate (microsatellite) to low (mitochondrial) gene diversity in dolphins using the Shannon Estuary and the Connemara–Mayo region, while dolphins that stranded along the coast showed much higher levels of gene diversity at both classes of markers (Mirimin *et al*. 2011).

Further south, a genetic study by Fernández *et al*. (2009b) of stranded bottlenose dolphins in Galicia, NW Spain, using mtDNA and microsatellites, found significant differences between animals in northern and southern Galicia, as well as from Portugal. There were also significant differences in microsatellite frequencies between southern Galicia and the northeastern corner of Spain. However, most of these sample sizes were very small. These studies, as well as one by Nichols e*t al.* (2009) using ancient DNA that found significant differentiation in a now extinct population from the Humber Estuary, Eastern England, suggest that local adaptation leading to isolation and potential extinction of coastal populations of bottlenose dolphin in Europe may be a feature of this species, exposing them to long-term conservation risk. Offshore populations, on the other hand, seem to exhibit much higher levels of gene flow (e.g. Madeira and the Azores – see Querouil *et al.* 2007).

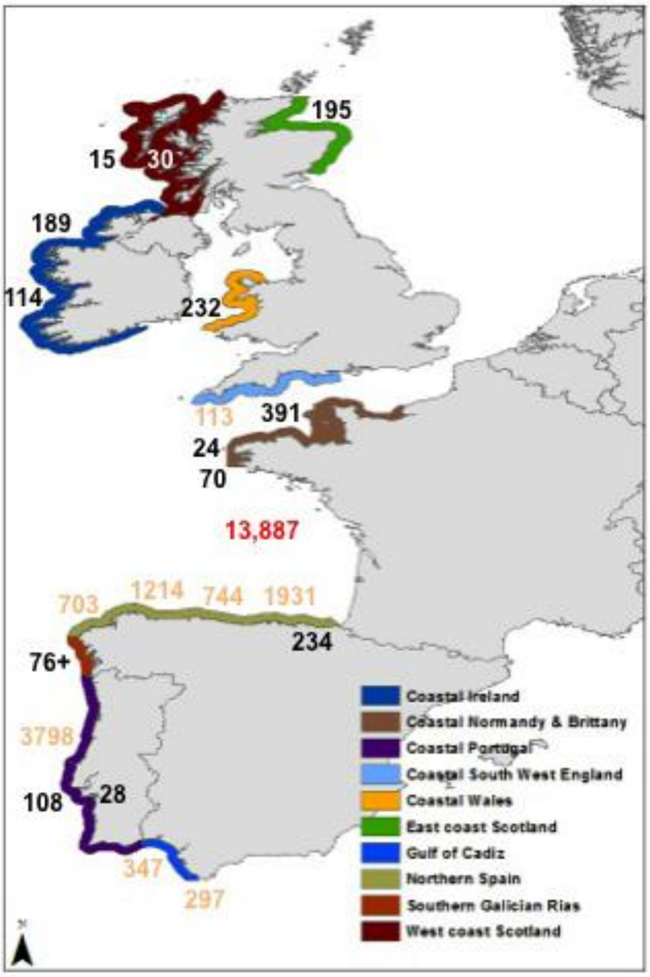

Figure 1.1.1 *Assessment Units and their population estimates for bottlenose dolphin in coastal Atlantic Europe. Values in red refer to populations believed to be largely of the offshore ecotype. Those in black are believed to be of the coastal ecotype, and the value in orange may be a mixture of the two.*

Conclusions

With our current knowledge, the offshore bottlenose dolphin ecotype in the Atlantic could be listed under category S4 a population with individuals with inherently large home ranges forming one mixing population at the supra-national level, or in category MR1 as it has a clearly sustainable metapopulation above Member State level and is not generally considered to be a fully migratory species.

The coastal ecotype forms a number of management units (at least ten) in Atlantic Europe (MATL biogeographic region of ES, FR, IE, PT and UK) and likely forms others in the Mediterranean and Black Sea. Some of these occur across Member State boundaries (e.g. Channel Islands and French Normandy coasts) but most occur below Member State level, and sometimes are very localised (e.g. Shannon Estuary, Ireland; and Sado Estuary, Portugal). These would therefore most closely fit either category S1, populations (assessment units) with more or less exchange at or below national level, or possibly in a few cases (e.g. Black Sea), category S2, populations of large, more or less mobile sedentary species with only one or a few clearly isolated population(s) within a Member State.

Step 1.3 Historical perspective: what happened to the species?
The common bottlenose dolphin was more widely distributed in the European mainland coastal zone in historical times (see Figure 1.1.2). Since the late 19th century, a number of coastal bottlenose dolphin populations, particularly those occupying estuaries, have declined or disappeared altogether. Bottlenose dolphins appear to have used some coastal areas for only limited periods of time, possibly forming ephemeral populations. For example, a group of dolphins utilised the Noirmourtier area (France) in the 1950-60s and similar reports have been made for the Quiberon-Houat-Hoedic area (France). It is unclear whether these were truly resident coastal populations or offshore visitors that remained in those areas for a limited period of time. The best recent example is use of the Marsdiep area and the area east of Texel (The Netherlands). The species was recorded there regularly, and in relatively large numbers (up to 30-40 at a time), between 1933 and 1939 by Verwey (1975), mainly between February and May, coinciding with the migration and spawning period of the Zuiderzee herring. After the closure of the Zuiderzee bay in 1932, the Zuiderzee herring gradually disappeared from the area, and in the late 1930s the regular occurrence of relatively large numbers of bottlenose dolphins ceased. Observations outside the Marsdiep area between the 1930s and 1970 are anecdotal, but the species was regarded as common in all Dutch waters and estuaries, second only to the harbour porpoise (Camphuysen & Peet 2006, Camphuysen & Smeenk 2016). After 1970, the species became scarce in Dutch waters, with strandings also declining rapidly (Kompanje 2001, Camphuysen & Peet 2006, Camphuysen & Smeenk 2016), at a similar time as a reduction in stranding records from SE England, as well as further west in SW England (Evans 1980, Tregenza 1992).

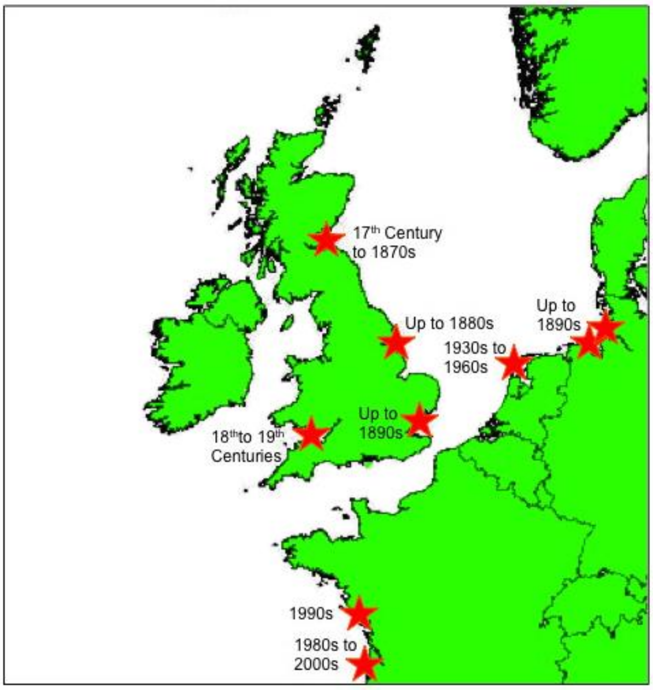

Figure 1.1.2 *Historical distribution of coastal bottlenose dolphins in NW Europe (from Evans & Scanlan 1990, ICES 2016)*

Earlier status changes are difficult to ascertain but historical accounts indicate that bottlenose dolphins occurred in the Severn Estuary, Thames Estuary, Humber Estuary, and Firth of Forth until the late nineteenth century (Evans & Scanlan 1990, Nichols *et al.* 2007). Along the coast of Germany, bottlenose dolphins occurred in the Elbe (Goethe 1983) and Weser estuaries (Mohr 1935, Goethe 1983, Kölmel & Wurche 1998) until the late nineteenth century. Further south, in Portuguese waters, bottlenose dolphins were reported in the Tagus estuary until 1960 (Teixeira 1979). More recently, a coastal group persisted at Arcachon (France) between the late 1980s until it disappeared in the early 2000s, and a group of six animals occurred at Pertuis Charentais, between Ré and Oléron Islands and the French mainland, for a period in the late 1990's.

In conclusion, currently, coastal bottlenose dolphins occur within each of the assessment units. In past centuries, the species occupied the southern North Sea and a number of estuaries where they now do not occur, or occur only as an uncommon visitor.

A number of human activities may affect bottlenose dolphins. The most obvious human pressure in coastal areas is human disturbance mainly from recreational activities (including commercial dolphin watching), with both short- and long-term impacts noted in several populations around the world (Bejder & Samuels 2003, Bejder *et al.* 2006) including West Wales (Feingold & Evans 2014a; Norrman *et al.* 2015) and East Scotland (Pirotta *et al.* 2014, 2015). Incidental bycatches of bottlenose dolphins through entanglement in fishing gear (mainly gillnets and pelagic trawls) also occur (ICES 2015a; b). Within the OSPAR region, bottlenose dolphin incidental bycatch appears to be highest (and potentially unsustainable) off the coasts of northern Spain (Galicia, Asturias, Cantabria and Basque Country), west Portugal, and SW Spain (Andalucia) (López *et al.* 2003, 2012, Goetz *et al.* 2014, Vázquez *et al.* 2014, Vélez 2014, ICES, 2015a). Fishing activities may also indirectly affect populations through depletion of the prey resource (ICES, 2015c). Habitat disturbance as a result of fishing activities causing damage to the seabed and its benthic faunas has been suggested as a human pressure in some areas, although this remains to be supported by evidence (Feingold & Evans, 2014a; Norrman *et al.*, 2015). Habitat loss has also affected coastal populations (Camphuysen & Peet 2006, Camphuysen & Smeenk 2016). Research has demonstrated high pollutant loads in most of the investigated coastal bottlenose dolphin populations, possibly leading to health issues and reproductive failure (Jepson *et al.* 2013, 2016). Exposure to high pollutant levels has been suggested as one reason for past declines and disappearance of some populations (Jepson *et al*. 2013, 2016). Climate change may also affect bottlenose dolphins, positively or negatively, either by altering human activities and thus pressures, or by affecting the stock sizes and distributions of their prey (Evans & Bjørge 2013), and increased underwater noise may have negative effects (Bailey *et al.* 2010).

Step 1.4 Analysis of distribution and trends

The only large-scale abundance surveys in Europe have been those of SCANS (July 1994) and SCANS II (July 2005) in NW European shelf seas, and CODA (July 2007) surveying offshore largely beyond the NW European shelf break. SCANS II surveys (2005) gave an overall abundance estimate of 12,600 (CV=0.27; 95% CI: 7,500-21,300) bottlenose dolphins, with highest densities near the shelf break (Hammond *et al*. 2013), whereas the CODA survey (2007) beyond the shelf edge yielded an abundance estimate (uncorrected for g(0) and responsive movement), of 19,300 (CV=0.25; 95% CI: 11,900-31,300) (CODA 2009). These highlight the significant offshore populations of this species. The coastal Atlantic Europe populations are thought to number somewhere between 2-4,000 individuals (ICES, 2016; Fig. 1.1.1). In July 2016, a further large-scale abundance survey, SCANS III was undertaken. Preliminary results have been published (ICES 2017) but are awaiting supplementary information to be made available from surveys in Irish waters (OBSERVE Programme) before complete results can be presented.

Of the available datasets, only five sites, all in the Atlantic region, have had sufficient repeated abundance estimates to allow any assessment of trends in abundance. For each assessment unit (AU), the estimated population size and trend, where known, are summarised below.

West coast of Scotland AU

A small resident bottlenose dolphin population numbering around 15 animals inhabits the vicinity of the Sound of Barra in the Outer Hebrides (Grellier & Wilson 2003, Cheney *et al.* 2013, Van Geel 2016) whilst an estimated 30 bottlenose dolphins range around the Inner Hebrides spending periods of time around Islay, the Small Isles, Skye, and occasionally the Minch north of Skye (Cheney *et al.* 2013, Van Geel 2016). There are insufficient data to determine the population trends at this time, although the numbers appear to be stable over the two last decades.

East coast of Scotland AU

Monitoring of bottlenose dolphins in the inner Moray Firth started in 1990, and later was extended to a wider part of the Firth. Even though bottlenose dolphins ranged all along the north and south coasts of the Moray Firth during the 1990s, it was not until the mid-1990s that the species started extending its range around the Grampian coast (Evans *et al.* 2003, Wilson *et al.*, 2004). It is now regularly encountered particularly off Aberdeen, the coast of Fife and in St Andrews Bay (Weir & Stockin 2001, Cheney *et al.* 2013). Bottlenose dolphins, some of which have been photo-identified as belonging to the Moray Firth population, are now seen annually along the coast of NE England as far south as Yorkshire (Sea Watch Foundation, unpublished data).

Analysis of mark-recapture studies using a Bayesian approach estimates the population on the East Coast of Scotland at 87-208 animals, with the latest estimate (2014) being 170 (95% Highest Posterior Density interval: 139-200). Despite inter-annual variability, the population is considered to be stable or increasing, with no decline over the available time series of >5% in 10 years (Figure 1.1.3; Cheney *et al.* 2013).

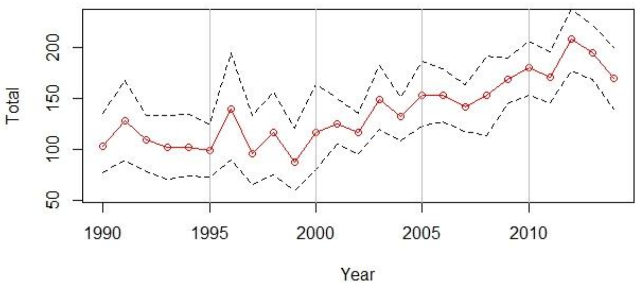

Figure 1.1.3 *Population trend, East Coast of Scotland AU*

Although bottlenose dolphins are occasionally recorded offshore in the North Sea and in coastal waters off SE England, northern France, Belgium, The Netherlands and Germany, there is no evidence that these are anything but transient animals, most likely from the East Coast of Scotland population or from further afield (Evans *et al.* 2003, Camphuysen & Peet 2006, Evans & Teilmann 2009, ICES 2013).

Coast of Wales AU

Annual monitoring of bottlenose dolphins in Cardigan Bay Special Area of Conservation, West Wales (UK), began in 2001. This was extended to incorporate the wider Cardigan Bay area from 2005. In addition, since 2007, there have been opportunistic photo-ID surveys in the coastal waters of North Wales, and occasionally around the Isle of Man and in Liverpool Bay (Pesante *et al.* 2008, Feingold & Evans 2014a, Norrman *et al.* 2015). A proportion of the population inhabiting Cardigan Bay in summer ranges more widely between November and April, occurring particularly off the north coast of Anglesey, the mainland coast of North Wales and further north around the Isle of Man (Feingold & Evans 2014b). Summer mark-recapture estimates for Cardigan Bay SAC have varied from 116-260 animals. The latest estimate (2015) is 159 (95% CI = 130-228) animals. For the wider Cardigan Bay (including both SACs), summer mark-recapture estimates have varied from 152-342 animals, with the 2015 estimate being 222 (95% CI: 184-300) animals. The Coastal Wales Assessment Unit population is considered to be stable, with no decline over the available time series of >5% in 10 years (Figures 1.1.4 & 1.1.5). It is noted that between 2013 and 2015, the population estimates have been amongst the lowest recorded but due to variability in the estimates it is too early to determine whether this indicates a decline.

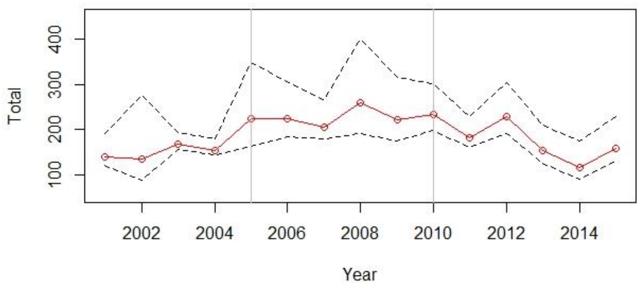

Figure 1.1.4 *Population trend, Cardigan Bay SAC (Coast of Wales AU)*

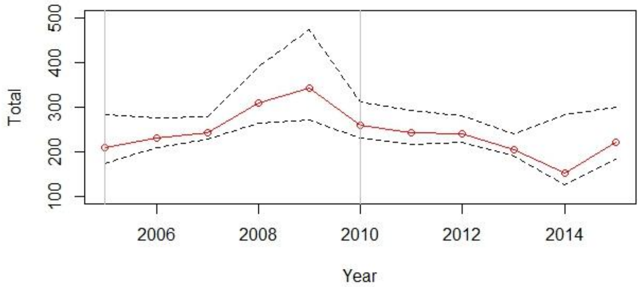

Figure 1.1.5 *Population trend, Wider Cardigan Bay (Coast of Wales AU)*

Coast of Ireland AU
Bottlenose dolphins are regularly recorded in several bays along the west coast of Ireland, notably Kenmare River and Brandon Bay (Co. Kerry), Clew Bay and adjacent coastal areas of Connemara (Co. Galway), Broadhaven Bay (Co. Mayo), and Donegal Bay (Co. Donegal) (Ingram *et al.* 2001, 2003, Ó Cadhla *et al.* 2003, Evans *et al.* 2003). They have also been recorded all along the south coast of Ireland, with sightings mainly around Cork Harbour (Co. Cork) and Rosslare Harbour (Co. Wexford) (Evans *et al.* 2003; O'Brien *et al.* 2009). Photo-ID matches indicate that individual bottlenose dolphins may range all around the coast of Ireland, and although there is a more or less continuous distribution from inshore to offshore, there is both photo-ID and genetic evidence for an offshore ecotype west of Ireland (O'Brien *et al*, 2009, Mirimin *et al.* 2011, Oudejans *et al.* 2015). There are a number of mark-recapture population estimates for animals using the west coast of Ireland, but at different spatial scales. One estimate for NW Connemara is 171 individuals (95% CI: 100-294) in 2009 (Ingram *et al*. 2009) and a second estimate for a much larger area, including Connemara, Mayo and Donegal, numbered 151 (95% CI: 140–190) individuals for the year 2014 (Nykanen *at al*. 2015). This mobile population appears to range widely, with seasonal and patchy habitat use. There is not enough information to indicate population trends.

Bottlenose dolphins inhabit the Shannon Estuary year-round, and genetic studies indicate that they form a discrete population separate from those occurring elsewhere along the west coast of Ireland (Mirimin *et al.* 2011). Six mark-recapture population estimates produced between 1997 and 2015 range from 107 to 140 individuals (Ingram 2000, Ingram *et al.* 2008, Berrow *et al*. 2010). The latest population estimate (2015) is 114 (95% CI: 90-143) (Rogan *et al*. 2015) indicating that the population is probably stable (Figure 1.1.6).

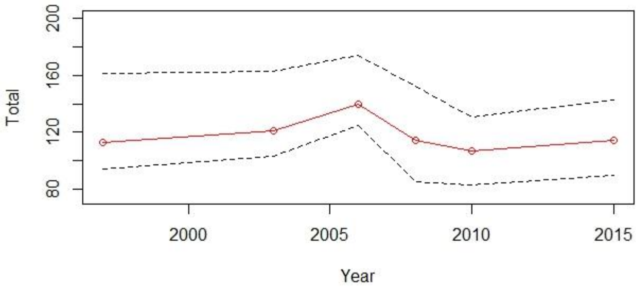

Figure 1.1.6 *Population trend, Shannon Estuary (Coast of Ireland AU)*

Coast of Southwest England AU
Bottlenose dolphins have regularly inhabited the south and southwest coasts of England since the 1990s, being most common around Cornwall but rare east of Dorset (Williams *et al.* 1998, Evans *et al.* 2003, Brereton *et al.* in review). No systematic photo-ID surveys have been undertaken, but Brereton *et al.* (in review) have reported maximum abundance estimates for south-west England coastal waters, using two mark-recapture methods, ranging between 102 and 113 (95% CI: 87-142) individuals over the combined period 2008-2013. There are insufficient data to assess trends.

Coast of Normandy and Brittany AU (France, UK)
A resident population of bottlenose dolphins inhabits the Gulf of St Malo, ranging between the French coast of Normandy and the Channel Islands (Couet 2015a, b, Louis *et al.* 2015). Mark-recapture estimates of this population in 2010 indicated it numbering between 372 (95% CI = 347-405) and 319 (95% CI= 310-327) animals, with a 2014 estimate of 340 (95% CI = 290-380) animals (Couet 2015a, b, Louis *et al.* 2015), thus indicating no significant difference (Figure 1.1.7).

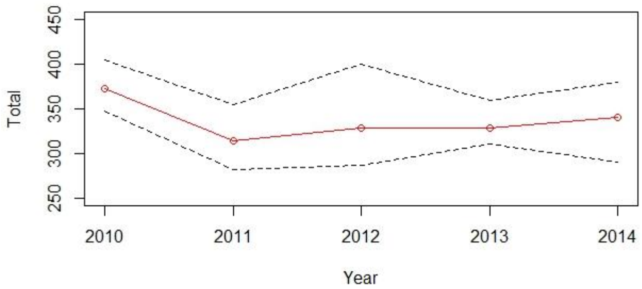

Figure 1.1.7 *Population trend, Gulf of St Malo (Coast of Normandy and Brittany AU)*

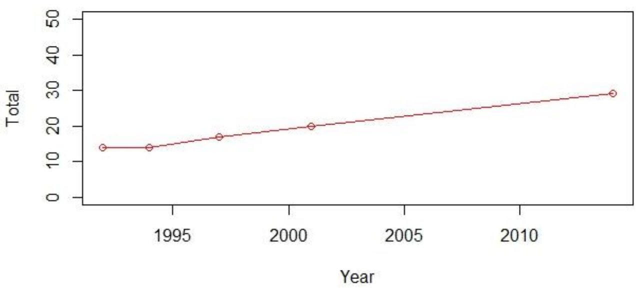

Figure 1.1.8 *Population trend for Ile de Seine (Coast of Normandy and Brittany AU)*

Two small populations, which appear to be distinct, exist in the Iroise Sea, one around Ile de Sein and the other around the Molène Archipelago. Photo-ID surveys have been undertaken in the vicinity of Ile de Sein since 2001, with at least five separate counts, ranging from 14 individuals in 2001 to 29 in 2014. An earlier estimate for this population was 14 animals in 1992 (Liret 2001, Liret *et al*. 2006), thus indicating a steady increase (Figure 1.1.8).

Around the Molène Archipelago, a mark-recapture estimate of 29 individuals (95% CI = 28-42) was produced from photographs taken between 1999 and 2001 (Le Berre & Liret 2001, Liret *et al.* 2006, Louis & Ridoux 2015). A new photo-ID analysis is currently being undertaken (V. Ridoux *pers. comm*.). It is therefore currently not possible to assess trends in this population.

Northern Spain
In northern Spanish waters, only model-based abundance estimates exist, derived from line-transect surveys conducted between 2003 and 2011. These encompass both coastal and offshore animals (López *et al.* 2013). The annual uncorrected abundance estimate in the study area is 10,687 individuals (95% CI of 4,094-18,132). Estimated abundances for the different areas are: (1) Euskadi: 1,931, (2) Cantabria: 744, (3) Asturias: 1,214, (4) Galicia: 703, (5) Galician Bank: 108 and (6) Aviles: 234. Although the distribution is homogeneous throughout the northern peninsula, there is a clear gradient in density, this being higher in eastern areas of the Bay of Biscay where the largest groups have been recorded (López *et al.* 2013). There are insufficient data at this time to make an assessment of trends.

Southern Galician Rias (North-west Spain)
Along the Galician coast, photo-ID surveys have been conducted between 2006 and 2009, resulting in the identification of 255 individuals (García *et al.* 2011). A third of these photo-identified individuals (n=76) were considered to form the resident population inhabiting the Southern Galician Rias, as revealed by recapture histories, genetics and stable isotope analysis (Fernández *et al.* 2011a, b, García *et al.* 2011). Movements of individuals were recorded between Galicia and Euskadi in the Bay of Biscay (García *et al.* 2011). It is not possible to make an assessment of trends in this population at this time.

Coast of Portugal
Bottlenose dolphins occur widely along the coast of Portugal as well as further offshore. Photo-ID surveys undertaken over two time periods have been used to derive mark-recapture population estimates of bottlenose dolphins in coastal Setúbal Bay (Martinho 2012, Martinho *et al.* 2015). Bottlenose dolphins identified from 1998-2001 were considered a closed and a more cohesive group than those from 2007-2011, with stable associations and an abundance of 106 (95% CI: 69–192) individuals. The more recent animals sampled seemed to be composed of an open group of 108 (95% CI: 83–177) animals, with a migration rate of 19% per year and low association values.

A wider-scale analysis of animals photographed in central west coastal Portugal from Nazaré and Sétubal Bay between 2008 and 2014 resulted in an estimate of 352 individuals (95% CI: 294-437) (Martinho 2012, Martinho *et al.* 2015).

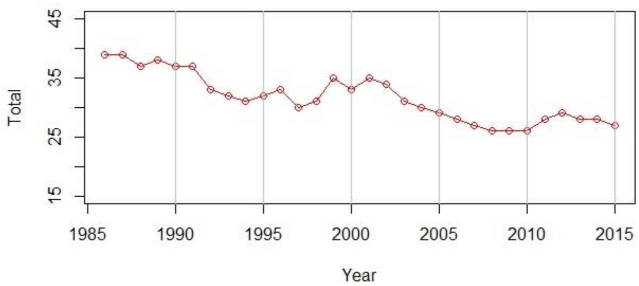

Figure 1.1.9 *Population trend, Sado Estuary (Coast of Portugal)*

There have been a number of line-transect surveys by both ship and plane undertaken off West Portugal between 2010 and 2014, covering the region between the coast and 50 nautical miles offshore. The estimated abundance for 2010-14 from aerial surveys was 2,306 animals (34.7% CV), and in 2011, using vessel surveys, it was 3,798 animals (87.6% CV) (Araújo *et al.* 2014). It is not possible to distinguish between the coastal population of this Assessment Unit and the offshore population.

The longest sequence of counts for a coastal bottlenose dolphin population in Europe is associated with the resident population in the Sado Estuary, where an annual census has been undertaken since 1986 (Gaspar 2003, Lacey 2015). Over this period, the population has shown a long-term decline from 39 individuals in 1986 to 28 individuals in 2014 (Figure 1.1.9), with pollution of the estuary proposed as a possible background for the decline (Van Bressem *et al*. 2003).

Gulf of Cadiz (Spain)
Mark-recapture estimates for bottlenose dolphins in the coastal Gulf of Cadiz have been determined for two periods: 2005-2006 and 2009-2010 (MAGRAMA 2012). These gave estimates of 347 individuals (95% CI: 264-503) for 2005-2006, and 397 (95% CI: 300-562) for 2009-2010 suggesting no significant difference. A much larger population apparently occupies the offshore Gulf of Cadiz, estimated to number 4,391 animals (95% CI: 2,373-8,356) in 2009-2010 (MAGRAMA 2012). It is not possible to make an assessment of trends at this time.

A bottlenose dolphin population also inhabits the area around the Strait of Gibraltar, on the edge of Area IV. Photo-ID surveys in 2010 resulted in a mark-recapture population estimate of 297 individuals (95% CI: 276-332) (Portillo *et al.* 2011). It is not possible to make an assessment of trends at this time.

Conclusions

In several areas within Europe (as has been found elsewhere in the world), coastal bottlenose dolphins are thought to form a number of demographically independent populations commonly referred to as management (or assessment) units. These have only been established within the Atlantic Region. Population trends for these should therefore be considered separately. Within the AUs, most of the available data relating to abundance (and therefore for examining trends) are from photo-identification studies of small localised resident or semi-resident groups, which are often related to monitoring the numbers of animals in protected areas. Evidence of trends was evaluated by examining abundance estimates at specific sites within the AUs where sufficient data exist (i.e., at least four abundance estimates from different years over a ten-year time period). No estimates exist for any site before 1986, and for most, estimates started only after the Habitats Directive came into force. Thus, there are no data prior to human impacts in these areas, so it is not possible to set a historical baseline. Although the historical abundance and distribution is unknown, there is good evidence that the species was once more widely distributed around these coasts. The recent trends available for six sites indicate little change over a ten-year period, with the exception of the small isolated population in the Sado Estuary, Portugal, which has shown a steady decline since the mid-1980s.

Step 2.1 FRP assessment

Virtually no quantitative information exists of population numbers for bottlenose dolphin before the Habitats Directive came into force in 1992, and it should be noted that this species is actually better known than most other cetaceans. Nevertheless, there is strong evidence that the species was more widely distributed (and therefore probably more abundant) in the European mainland coastal zone, with regular occupation of a number of estuaries in historical times (see step 1.3). The indication therefore is that FRP for several coastal assessment units should be above the current values. The common argument used for setting FRPs at larger values to those obtained from recent abundance estimates is that environmental conditions have changed since so that the carrying capacity for a local population may also be lower than any FRP assessment. On the other hand, there is evidence that if conditions are improved, areas can be re-colonised (as, for example, appears to be occurring around the Firth of Forth, East Scotland). With some understanding of habitat preferences and food resource needs, however, one can compare against relatively undisturbed extant populations, and use that information as a baseline. This would best be undertaken by assessment unit where possible.

For the offshore ecotype, one could use the best estimate of Current Value as the FRP. For the coastal ecotype and the various assessment units within it, historical baselines would be preferable. These would best be derived by genetic analysis to estimate Ne, genetic diversity, population history (evidence of bottlenecking), and degree of connectivity between AUs. The order of magnitude of FRP cannot be determined until the genetic analyses described above have been undertaken. It is likely that the Ne/N ratio for coastal bottlenose dolphin is less than has been applied for mainly terrestrial species, but in any case, levels of connectivity between populations need to be better assessed first.

Conclusions

For the offshore ecotype within MATL FRP=CV

For the coastal ecotypes within MATL FRPs for the assessment units (AUs) are unknown but can be derived by genetic analysis to estimate Ne (effective population size), genetic diversity, population history (evidence of bottlenecking), and degree of connectivity between AUs. 'Guestimates' what the FRV might be for each AU in terms of broad ranges of population estimates are:

West coast of Scotland AU: 50-500
East coast of Scotland AU: 150-500
Coast of Wales AU: 250-500
Coast of Ireland AU: 300-1000 (Shannon Estuary: 100-200)
Coast of Southwest England AU: 100 -300
Coast of Normandy & Brittany AU: 500-1000
Sado Estuary AU: 30-50

Step 2.2 FRR assessment

As described earlier, coastal bottlenose dolphin populations once existed in most of the major estuaries and bays of Western Europe. The degree to which they were permanent residents is difficult to ascertain historically, although in the last century, some areas (e.g. southern North Sea) had populations regularly living there for periods lasting at least some decades. Given that the species is highly mobile, there is potential for the locations currently without bottlenose dolphin populations, to be re-colonised, so long as there are suitable environmental conditions to sustain adequate prey resources. As a good example, the population normally inhabiting the Moray Firth in Northeast Scotland has been extending its range along the coast since the 1990s (ICES, 2016). In the last ten years, bottlenose dolphins identified as from the Moray Firth population have been seen annually in Eastern England, and in summer 2016, the species was even recorded off the coast of Norfolk in the southern North Sea. Up to now, these have been transient movements, but if conditions were suitable, there is potential for the species to re-occupy the southern North Sea as a resident.

Conclusions

For the offshore ecotype within MATL FRR=CV

For the coastal ecotypes within MATL FRRs for the assessment units (AUs) are unknown but can be assessed when FRPs are available.

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1.2 Short-beaked common dolphin (*Delphinus delphis*) in Europe

Step 1.1 Biology of the species

The short-beaked common dolphin (*Delphinus delphis*) is one of the smallest of the true dolphins (family Delphinidae). It is an active, fast-moving species, frequently bow-riding boats and jumping clear of the water. Two species of common dolphin are currently recognised in the North Atlantic – the short-beaked form, *D. delphis*, and the long-beaked form, *D. capensis,* but only the former has been recorded in European seas. It is a social species, usually travelling in schools of 6-15 individuals, but can form aggregations numbering hundreds or even low thousands, associated with feeding or large-scale movements. There is evidence for some segregation by age and sex. Food-herding behaviour is frequently observed with apparent cooperation between school members. Mating and calving are seasonal, largely between May and September. Males reach sexual maturity around 11-12 years, females around 9-10 years of age. On reaching sexual maturity, females give birth to a single calf, every 3-4 years. Maximum age reported in the North-east Atlantic is 28 years. Generation length has been estimated at 14.1 years (14.8 years in populations at a stable state). Common dolphins appear to be opportunistic feeders, but most common prey recorded in the region are epi- and mesopelagic fish including horse mackerel, mackerel, Norway pout, sardines, anchovy, whiting, scad, sprat, sandeel, and blue whiting, as well as small cephalopods. More detailed accounts of the biology of the species can be found in Evans (1994), Heyning & Perrin (1994), Murphy *et al*. (2008, 2013), and Perrin (2009).

Step 1.2 Spatial scale of functioning

The short-beaked common dolphin has a worldwide distribution in oceanic and shelf-edge waters of tropical, subtropical and temperate seas, occurring in both hemispheres. It is abundant and widely distributed in the eastern North Atlantic, mainly occurring in deeper waters from Macaronesia and north-west Africa north to approximately 65ºN latitude (although rare north of 62º N), west of Norway and the Faroe Islands (Reid *et al*. 2003, Murphy *et al*. 2008). It occurs westwards at least to the mid-Atlantic ridge (Doksaeter *et al*. 2008, Cañadas *et al.* 2009), and eastwards it enters the western Mediterranean, with a distinct isolated population in the Black Sea (Bearzi *et al*. 2003) (Fig. 1.2.1). Sightings are rare in the eastern section of the English Channel and the southern North Sea (Evans *et al*. 2003, Camphuysen & Peet 2006). In the offshore North Atlantic it seems to favour waters over 15º C and shelf edge features at depths of 400-1,000m between 49º-55º N especially between 20º-30º W (Forcada *et al.* 1990, Cañadas *et al.* 2009).

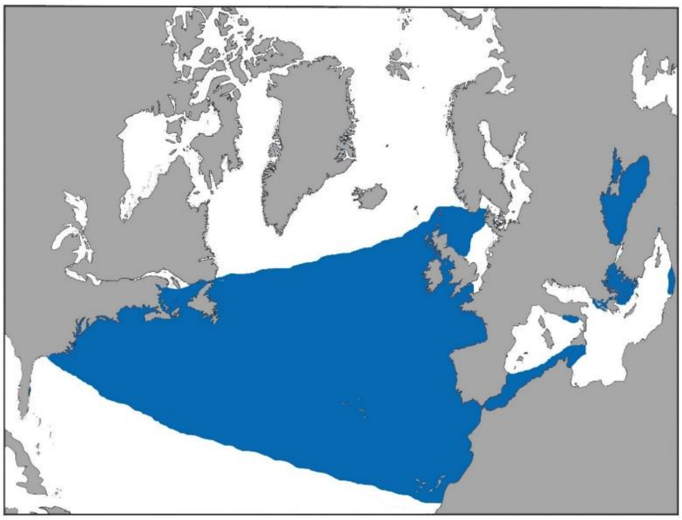

Figure 1.2.1 *North Atlantic distribution of short-beaked common dolphin (depicting those areas where the species is thought to regularly occur)*

On the UK continental shelf, the species is common in the western half of the English Channel and the southern Irish Sea, and further north in the Sea of Hebrides and southern part of the Minch (Evans *et al*. 2003, Reid *et al*. 2003). It is also common south and west of Ireland. In some years, the species occurs further north and east in shelf seas - in the northern Hebrides, around Shetland and Orkney, and in the northern North Sea. It is generally rare in the central and southern North Sea and eastern portion of the English Channel, but is abundant in the Bay of Biscay (Evans *et al*. 2003, Reid *et al*. 2003, Kiszka *et al.* 2007, Laran *et al*. 2017). Range shifts may be related to changing oceanographic conditions (warmer sea temperatures and extensions of the range of potential prey species like anchovy and sardine). Since the 1990s, the species has become regular in the northern North Sea and even entered the Baltic (Evans *et al*. 2003, Murphy *et al*. 2008, Kinze *et al*. 2010).

In shelf waters off the west coasts of Ireland and Scotland, and in the Irish Sea, common dolphin abundance tends to be greatest in the summer months (Evans *et al*. 2003, Berrow *et al*. 2010), whereas further south, the western English Channel and Bay of Biscay remain important winter habitats for the species, which is abundant there between December and March (Brereton *et al.* 2005, De Boer *et al*. 2008), although abundance in the Bay of Biscay is greatest in the summer (Laran *et al*. 2016). The species also occurs all along the coast of Portugal and southwest Spain.

Although common dolphins can be found over large parts of the eastern North Atlantic and appear to form one large panmictic population (Mirimin *et* al. 2009, Amaral *et al*. 2012, Moura *et al*. 2013), there is also some suggestion for the existence of separate populations, e.g. one occurring primarily offshore and the other over the continental shelf, although this needs further clarification (Caurant *et al*. 2009, Evans & Teilmann 2009). Genetic (mtDNA) studies indicate a significant level of divergence between Mediterranean (Alboràn Sea) and Atlantic populations (Natoli *et al*. 2008), although with directional estimates of gene flow suggesting some movement of females out of the Mediterranean (possibly due to oceanographic features such as the Almería-Orán thermohaline front). Differences in contaminant levels between dolphins from the Alboràn Sea and Atlantic Ocean also suggest a certain degree of isolation (Borrell *et al*. 2001).

Although common dolphins are relatively abundant in the westernmost portion of the Mediterranean basin - the Alboràn Sea, the species is rare or completely absent from large parts of the Mediterranean (Bearzi et al. 2003). Mediterranean regions where common dolphins have apparently vanished include the Adriatic Sea, Balearic Sea, Provençal basin, and Ligurian Sea, the Saronic Gulf, and the Dodekanese (Frantzis et al. 2003). There are only sparse records with possibly isolated groups present around Sardinia and Corsica (Bearzi et al. 2003), and seasonally in the south-eastern Tyrrhenian Sea off the island of Ischia (Mussi et al. 2002). The species is present in the Sicily Channel and around Malta (Vella 2002), in portions of the eastern Ionian Sea, particularly around the island of Kalamos (Politi & Bearzi 2001), and in the Gulf of Corinth (Frantzis & Herzing 2002). Genetic studies show differentiation between the western and eastern Mediterranean populations, and suggest that a population bottleneck likely occurred as a result of a sharp population decline in the recent past (Natoli et al. 2008, Moura et al. 2013).

At the eastern end of the Mediterranean, there is little indication of movement by common dolphins through the narrow Dardanelles Strait between the Aegean and the Marmara and Black Seas, where common dolphins are known to occur (Öztürk & Öztürk 1997, Frantzis et al. 2003). A preliminary study of skull morphometrics (Amaha 1994) suggested differences between Black Sea and Mediterranean common dolphins. However, a genetic comparison of relatively small samples (8 Black Sea, 20 central Mediterranean) revealed no significant differences (Natoli et al. 2008). Clearly, further work based on larger samples is needed to assess and characterize the relationship between Black Sea and Mediterranean common dolphins. Some genetic exchange may occur in portions of the Aegean Sea where favourable habitat still exists (e.g., in the Thracian Sea; Frantzis et al. 2003) but what remains between the Aegean and Alboràn sectors of the Mediterranean seems to be only isolated, remnant groups (possibly indicative of further population substructure). The Black Sea population therefore remains classified as a separate subspecies *D. delphinus ponticus*.

Conclusions

Current knowledge indicates that the short-beaked common dolphin in the Atlantic and North Sea should be listed under category MR4, a population with individuals having inherently large home ranges forming one mixing population, or in category MR1 as it has a clearly sustainable meta-population above Member State level and is not generally considered to be a fully migratory species. The populations in the Alboràn Sea (western Mediterranean) and Aegean Sea (eastern Mediterranean) are geographically isolated and genetically distinct, as may be the one in the Black Sea. Since these fragmented populations nevertheless exist above Member State level, they would most closely fit category S4 as well.

Step 1.3 Historical perspective: what happened to the species?

There is little historical information from the European Atlantic to indicate trends in common dolphin abundance. The species has occurred in variable frequency in the North Sea with no particular trend (Evans et al. 2003, Reid et al. 2003, Camphuysen & Peet 2008), whereas it has been only infrequent in the Danish Belt Seas and Baltic (Kinze 1995). Once one of the most abundant species in the Mediterranean Sea, the common dolphin has experienced a generalized and major decrease in this region since the middle of the twentieth century (Bearzi et al. 2003). Dramatic declines were recorded in portions of the central Mediterranean, particularly in the northern Adriatic Sea and in the eastern Ionian Sea (Bearzi et al. 2004, 2006).

In the Black Sea, common dolphin populations collapsed by the mid 1960s following many years of overexploitation, which involved the killing of several hundreds of thousands of common dolphins in the mid-20th century (IWC 1983, Birkun 2006, 2008). Commercial killing of Black Sea common dolphins, as well as other Black Sea cetaceans, was banned in 1966 in the former USSR, Bulgaria and Romania, and in 1983 in Turkey. Since then, the population has shown little sign of recovery (Birkun 2006, 2008). It was calculated that between 1962 and 1966, the cetacean fishery in the USSR and Bulgaria landed 121,395 common dolphins, while during the preceding 31 years (1931-61) a further 1,449,304 had been landed mainly by the USSR (Zemsky 1996). Between 1976 and 1981, common dolphin was believed to account for 15-16% (or 37,500-40,000 individuals) of the Turkish catch, estimated for that period as 250,000 animals of all three species (common dolphin, harbour porpoise, and bottlenose dolphin) (IWC 1983).

In the European Atlantic, the main threat to common dolphins is probably bycatch, involving French, Spanish, Portuguese, Irish and UK pelagic trawl fisheries targeting a range of fish including albacore tuna, sea bass, blue whiting, hake, horse mackerel, sardine or anchovy (Tregenza et al. 1997, Tregenza & Collet 1998, Morizur et al. 1996, 1999, 2014, ICES 2016). Annual bycatch levels have been estimated in the hundreds or low thousands, from independent observer programmes, although these have not comprehensively assessed all fisheries. During the 1990s, the tuna driftnet fishery also caught large numbers of common dolphins until a ban was introduced in 2002 (Goujon et al. 1993, Goujon 1996, Rogan & Mackey 2007). Bycatches are also known to occur in gill nets, tangle nets and possibly other fisheries (Tregenza et al 1997, Cosgrove & Browne 2007). From post mortem examinations of 537 common dolphins stranded in the UK between 1991 and 2010, 51% were diagnosed as by-catch, 18% live-stranded, 7% from infectious disease, and 4% died from starvation (Deaville & Jepson 2011).

In the Mediterranean, incidental mortality in fishing gear, particularly driftnets, is thought to have contributed to the decline of common dolphins, although reduced availability of prey caused by overfishing and habitat degradation, contamination by persistent chemicals resulting in immunosuppression and reproductive impairment, and environmental changes such as increased water temperatures affecting ecosystem dynamics are also thought to have played a role (Bearzi et al. 2003, 2006). The Moroccan driftnet fishing fleet has been estimated to have an annual bycatch of c. 12,000-15,000 dolphins around the Strait of Gibraltar (Tudela et al. 2004).

In the Black Sea, besides the very large numbers of common dolphins hunted in the last century until bans came into force in the 1960s-80s, reduced prey availability (e.g. of anchovy and sprat) has been considered a major threat common dolphins (Bushuyev 2000). This was thought to contribute to the two mass mortality events that killed large numbers of common dolphins in 1990 and 1994 (Krivokhizhin & Birkun 1999), in the case of the latter linked to increased susceptibility to morbillivirus infection. Prey reduction is thought to have been caused by a combination of overfishing, eutrophication and the explosive increase of the introduced ctenophore *Mnemiopsis leidyi* (Birkun 2006). Prey depletion caused by overfishing was also considered as a main cause for the decline of common dolphins in the eastern Ionian Sea (Bearzi et al. 2006).

Step 1.4 Analysis of distribution and trends.

There have been several abundance surveys of common dolphin in various parts of the eastern North Atlantic though none spanning the entire region (e.g. MICA survey in 1993 - Goujon et al. 1993; ATLANCET aerial survey in 2001 - Ridoux et al. 2003; MARPRO surveys in 2007-12). In NW Europe, the SCANS survey (July 1994), covering an area from the Celtic shelf to c. 11°W and 48°S, produced an estimate of 75,449 individuals (CV=0.67; 95% CI: 23,900–248,900) (Hammond et al. 2002). However, this survey did not use a double-platform method, nor correct for animals missed on the track line, nor, perhaps most importantly, responsive movement – common dolphins notoriously respond positively to the presence of vessels that they can bow ride. In 2005, SCANS-II re-surveyed the same area as SCANS, but extended this to include also the Irish Sea, the waters off western and Northern Ireland, west Scotland, and continental shelf waters off France, Spain and Portugal. The total summer abundance for those Northeast Atlantic shelf waters was c. 56,221 (CV=0.23; 95% CI: 35,750–88,400) (Hammond et al 2013). This was supplemented by the CODA offshore survey conducted in July 2007 along the shelf edge of the ASCOBANS Agreement Area, which estimated a total abundance of 116,709 (CV=0.34; 95% CI: 61,400–221,800) (CODA 2009). These surveys corrected for animals missed along the track-line but not for responsive movement. The only analysis in the region that has not only corrected for animals missed along the track-line but also for responsive movement was for the western block of the 1995 NASS survey that covers part of the central North Atlantic (52º–57.5º N, 18º–28º W), and resulted in an estimate of 273,000 common dolphins (CV=0.26; 95% CI: 153,000–435,000) (Cañadas et al. 2009).

In July 2016, SCANS-III survey was undertaken and yielded an abundance estimate of 467,673 (CV=0.26; 95% CI: 281,100-778,000). This estimate excludes results from western and southern Ireland which are not yet available. The most striking difference is the much larger estimates of this species and striped dolphin (together totalling almost a million animals) in 2016 compared to 2005/07, nevertheless consistent with results from the French SAMM surveys in French waters of the Bay of Biscay and the Channel in summer 2012, which totalled almost 700,000 common and striped dolphins (Laran et al. 2017). Notwithstanding this, the lack of estimates for Irish waters in 2016 precludes any robust comparisons of estimates of abundance between 2005/07 and 2016 for the whole area.

In the western Mediterranean, abundance has been estimated at 19,400 (95%CI: 15,300-22,800) in the northern Alborán Sea between 2000 and 2004 (Cañadas & Hammond 2008). The size of the eastern Mediterranean population has not been estimated.

The population size in the Black Sea is unknown. Line transect surveys have been conducted recently to estimate common dolphin abundance in a few parts of the range. The survey areas are small relative to the total range of the subspecies. Results suggest that current population size is at least several tens of thousands, and possibly 100,000 or more (Birkun 2006).

Conclusions

The common dolphin was much more widespread in the Mediterranean in the middle of the last century, and in the Black Sea a crash in numbers was documented between the 1940s and the 1960s. Declines in the Mediterranean were thought to be caused by a combination of heavy bycatch and prey depletion. One obvious cause of the declines in the Black Sea was the excessive hunting of the species by several countries. That came to an end in the 1980s but since then the Black Sea population has shown little sign of recovery, probably because it is still experiencing prey depletion from overfishing and habitat degradation from pollution. If these human pressures were reduced, there is scope for recovery of depleted populations.

Common dolphins in the eastern North Atlantic appear to be in a healthier state although they continue to suffer fisheries bycatch in particular.

As with other cetacean species, long-terms trends in common dolphin numbers are generally lacking. There are no data on population sizes for the species in Europe prior to human impacts, so it is not possible to set a historical baseline. However, there is good evidence to indicate major declines over a large part of the Mediterranean and in the Black Sea. The three (1994, 2005/07, 2016) wide-scale July surveys in the European Atlantic are difficult to interpret because coverage has varied between surveys, and none has encompassed the entire eastern North Atlantic range. Nevertheless, they do not suggest a decline, and the variation observed may simply reflect differential movement between regions.

Step 2.1 FRP assessment

In the case of common dolphins in the European Atlantic, for which there is no historic evidence before 1992 when the Habitats Directive came into force but no obvious contraction in range or declines in orders of magnitude of abundance, it probably makes sense to treat FRP as equivalent to CV. For the Mediterranean and Black Sea, that is clearly not the case, the former having experienced a major contraction in range, and the latter in population size. In those regions, FRP should greatly exceed CV.

Whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

Conclusions

Short-beaked common dolphin and its close relative long-beaked common dolphin are amongst the most abundant of dolphin species globally. The order of magnitude of FRP in European seas is likely to be in the range of one to two million animals, but for separate populations, it will be correspondingly smaller. The major declines that have occurred historically in the Mediterranean and Black Sea could be reversed if conditions allowed.

FRP = CV (European Atlantic, MATL and MMAC)
FRP > CV (Mediterranean, MMED)
FRP >> CV (Black Sea, MBLS)

Step 2.2 FRR assessment

Common dolphins are wide ranging in the North Atlantic both on and off the shelf. The species also occurs in the Mediterranean and Black Sea, although in the case of the former, its range appears to be significantly reduced. Its current range at least in the Mediterranean could therefore likely expand if suitable conditions prevailed.

Conclusions
FRR = CV (European Atlantic, MATL and MMAC)
FRR >> CV (Mediterranean, MMED)
FRP > CV (Black Sea, MBLS)

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1.3 Harbour porpoise (*Phocoena phocoena*) in Europe

Step 1.1 Biology of the species

The harbour porpoise (*Phocoena phocoena*) is the only member of the family Phocoenidae occurring in Europe. It is a small, relatively cryptic species that tends to avoid vessels, and rarely bow-rides. It usually occurs solitarily or in small groups of 2-3. Where food is concentrated or when making long-distance movements, larger temporary aggregations may form, numbering one hundred or more. Group composition appears to be fluid, except for adult females with dependent calves. Young are born mainly between May and August (peak in June), although some as early as March. The mating season is between April and September, with a peak in July-August. Porpoises reach sexual maturity at 3-5 years, with little difference between sexes. Once sexually mature, females may give birth to a single calf every 1-2 years. Longevity is relatively short, usually up to 12 years, although the maximum is 24 years in both sexes. Generation length has been estimated at 8.3 years (11.9 years in populations at a stable state). Diet varies both geographically and seasonally, and comprises small demersal, bentho- or epipelagic fish, such as whiting, sandeel, sprat, herring, cod, pouts and gobies. More detailed accounts of the biology of the species can be found in Bjørge & Donovan (1995), Donovan & Bjørge (1995), Read (1999), Evans *et al*. (2008), and Read & Tolley (2009).

Step 1.2 Spatial scale of functioning

The harbour porpoise is found in temperate and sub-arctic seas of the northern hemisphere, occurring in both the Atlantic and Pacific. In the North Atlantic, the species occurs mainly from Central West Greenland and Novaya Zemlya in the north to North Carolina and Senegal in the south, with a geographically isolated population in the Black Sea (Reid *et al.* 2003, Evans *et al*. 2008, Fontaine *et al.* 2007, 2010). Although porpoises can be found in deep Atlantic waters off the edge of the continental shelf (for example within the Faroe Bank Channel), they are comparatively rare in waters exceeding 200 metres. The species frequently uses tidal conditions for foraging (see e.g., Evans 1997, Johnston *et al*. 2005, Pierpoint 2008, Marubini *et al*. 2009, Isojunno *et al.* 2012, Jones *et al*. 2014).

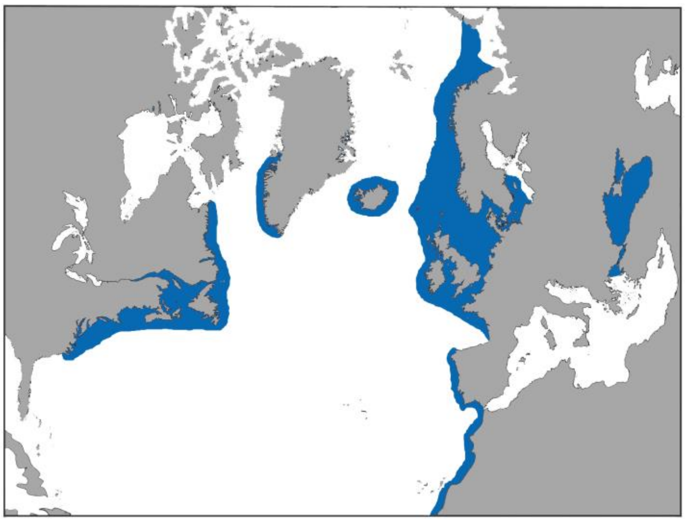

Figure 1.3.1 *North Atlantic Distribution of Harbour porpoise (depicting those areas where the species is thought to regularly occur)*

In European seas, it is common and widely distributed over the continental shelf (mainly at depths of 20-200m) from the Barents Sea and Iceland south to the coasts of France and Spain (Figure 1.3.1), although in the 1970s it became scarce in the southernmost North Sea, English Channel, and Bay of Biscay. Nevertheless, it remains the most widely distributed and frequently observed cetacean in North West European shelf seas, and since the 1990s, has returned to the southernmost North Sea, English Channel and French Biscay coast (Camphuysen 2004, Kiszka *et al*. 2004, 2007, Evans 2010, Hammond *et al*. 2013). It is largely absent from the Mediterranean (Frantzis *et al*. 2001), with small numbers reported mainly from the northern Aegean Sea, which probably come from the Black Sea population (Birkun & Frantzis 2008, Notarbartolo di Sciara & Birkun, 2010, Tonay & Dede 2013).

The isolated Black Sea harbour porpoise population is clearly genetically differentiated from Atlantic populations and has been classified as a separate subspecies, *Phocoena phocoena relicta* (Fontaine *et al*. 2007, 2010, 2017), but there is also genetic evidence that the Iberian population is distinct from porpoises further north; these are believed to have derived from an expansion of a small population off the northwest African coast (Fontaine *et al*. 2007, 2010). Porpoises along the French Atlantic coasts appear to be an admixture from the two genetically distinct populations along the Iberian coasts and in the North East Atlantic (Alfonsi *et al*. 2012). Further north, porpoises from southwestern UK appear to be differentiated from those in the North Sea (Fontaine *et al.* 2017).

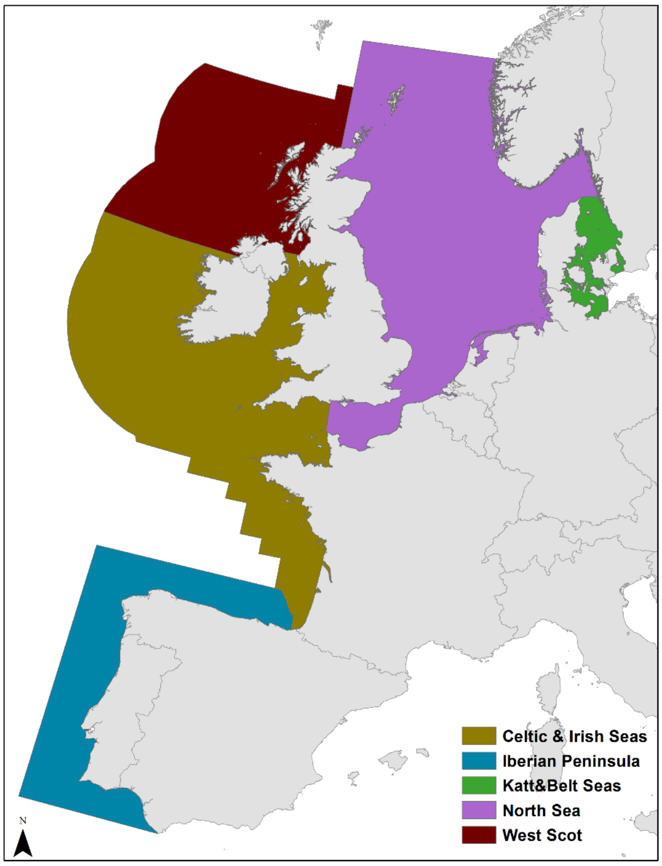

Figure 1.3.2 *Assessment Units for harbour porpoise in coastal Atlantic Europe (from ICES WGMME, 2013)*

Up to 16 separate Management Units (MU), thought to be demographically if not genetically distinct, have been proposed for the porpoise in the North Atlantic, using a precautionary approach of division rather than lumping where there is evidence to suggest this (Evans & Teilmann 2009, ICES 2012, Sveegaard *et al.* 2015, Fontaine *et al*. 2010, 2017, Lah *et al.* 2016). These include up to 11 MUs across North West Europe. These are: 1) the inner Baltic Sea (with its western boundary being around the Darss/Gedser underwater ridge or Rügen); 2) the southern Belt Sea; 3) the northern Belt Sea and southern Kattegat; 4) the northern Kattegat and Skagerrak; 5) the NE sector of the North Sea; 6) the western sector of the North Sea including the Eastern Channel; 7) the Celtic Sea along with South-west Ireland, the Irish Sea & Western Channel; 8) Northwest Ireland and Western Scotland; 9) the waters around the Faroe Islands south towards the Faroe-Shetland Channel; 10) the west French part of the Bay of Biscay; and 11) the Iberian Peninsula (coasts of Portugal and Atlantic Spain). The ICES Working Group on Marine Mammal Ecology (2014) has combined some of these into five assessment units (Figure 1.3.2). Further research is likely to refine and modify these subpopulations.

Various lines of evidence suggest that porpoises in the North Sea also exhibit some sub-structuring (Yurick & Gaskin 1987, Walton 1997, Lockyer 1999. Das *et* al. 2003, 2004, Teilmann *et al*., 2008; Sveegaard *et al*., 2011, De Luna *et al*. 2012). In particular, porpoises from the southwestern North Sea appear to differ significantly from those in the northeastern North Sea. There may be further sub-structuring between northern and southern North Sea and western vs eastern North Sea but it is difficult to determine where the division may lie given that different authors have used different sampling divisions and the precise origins of the samples are not necessarily known. Most studies indicate greater philopatry for females than male porpoises.

A number of authors allude to differences in ecology between animals from the northeastern and southern/western North Sea, particularly with respect to feeding. In recent years, seascape genetics has been used increasingly to account for differentiation observed between cetacean populations. In the absence of actual physical barriers, it has been shown in studies of various cetacean species that differences in ocean current systems and/or bathymetry parallel observed genetic differences. In the North Sea, the northeastern sector (from Shetland east to northern Denmark) is significantly deeper than the central and southernmost sectors. Current systems also show differences between the two regions both in terms of strength and circulation patterns. The central part of the North Sea represents an intermediate zone between the two regions, with respect to both bathymetry and ocean circulation.

If porpoises in the northeastern North Sea are feeding mainly upon pelagic prey (for which skull characteristics, particularly of the buccal cavity, have developed, as found by De Luna *et al*. 2012) whilst those in the southernmost North Sea are taking fish primarily off the bottom (with equivalent changes to the size of the buccal cavity), then these may be best considered separate management units with a potential boundary following bathymetric and oceanographic changes.

De Luna *et al*. (2012) and Andersen *et al*. (2001) also found significant differences between porpoises from the British North Sea and those from the Danish North Sea, as well as differences between porpoises from Norway and both the Danish North Sea and the British North Sea. And Wiemann *et al.* (2010) showed significant sub-structuring between the Danish North Sea and Norway. Thus, the presence of three Management Units within the North Sea may also be a possibility.

Within Danish waters, Wiemann *et al*. (2010) found clear genetic evidence of a population split between the Skagerrak and the Belt Seas, with a transition zone in the Kattegat area. This was particularly evident in significant frequency shifts of the most abundant mitochondrial haplotypes. A particular haplotype almost absent in the North Sea was the most abundant in the Belt Seas and Inner Baltic Sea. Microsatellites yielded a similar pattern.

Separation of the North Sea and the Baltic Sea populations with an east–west border within a transition zone in the Kattegat (waters South of the 58°N and North of 56°N latitudes) is supported by 3D skull geometric morphometric measurements (Galatius *et al*. 2012), satellite tracking (Sveegaard *et al*. 2011), as well as microsatellite and mitochondrial genetic analysis (Andersen *et al.* 2001, Wiemann *et al.* 2010). A combination of these studies suggests a transition zone between the Baltic Sea and Baltic Proper populations between Fehmarn Belt and 14°20’E (Sveegaard *et al.* 2015). The genetic evidence for a division between the Baltic Sea and the Baltic Proper populations is not as strong as the division between the North Sea and the Baltic Sea population, but Wiemann *et al.* (2010) nevertheless advocated a precautionary division into two separate Management Units. The less pronounced genetic and morphological differences may be due to the young age of the Baltic Proper population: a recent investigation indicates immigration and establishment of harbour porpoises in the Baltic Sea around 9000 years ago when the previously closed connection to the Atlantic Ocean was established through the Danish straits as a result of ice melting in the Arctic, following the last Ice Age (Sommer *et al*. 2008).

Most recently, Lah *et al.* (2016) undertook a genome-wide analysis, using single nucleotide polymorphisms (SNPs), derived from double digest restriction-site associated DNA sequencing (ddRAD-seq), as well as 13 microsatellite loci and mitochondrial haplotypes. They observed a distinct separation of the North Sea harbour porpoises from the Baltic Sea populations, but also identified splits between porpoise populations within the Baltic Sea. In particular, they observed a notable distinction between the Belt Sea and the Inner Baltic Sea sub-regions, supporting other authors who have advocated for porpoises in the Baltic Proper to be considered a distinct separate population.

From telemetry studies and passive acoustics, seasonal migrations of porpoises have been reported for the Skagerrak populations, which migrate westward in the winter towards the southern tip of Norway and into the northern North Sea, for the Kattegat populations which may migrate southward through the Great Belt in the winter, as well as for the porpoises from the Belt Sea and inner Baltic Sea, which may migrate westward to the Pomeranian Bay and from the Kadet trench into Danish waters, respectively (Verfuß *et al.* 2007, Sveegaard *et al*. 2011, Gallus *et al*. 2012, Benke *et al.* 2014).

<u>Conclusions</u>
Harbour porpoises are distributed more or less continuously across the shelf seas of northern Europe. Nevertheless, there is evidence (e.g. genetic, morphometric, telemetry studies of movements) for demographically distinct populations which we commonly refer to as management (or assessment) units. Animals around the Iberian Peninsula appear to be more closely related to those from NW Africa; in the Bay of Biscay the species is relatively uncommon and forms an admixture of animals from both south and north; the population in the Celtic Sea shows differences from those in the North Sea where there may be further sub-structuring, and separation from those in Danish Belt Seas. The porpoise population in the Baltic Proper is small, and distinct from those further west. The Black Sea population is geographically isolated from those in the Atlantic, the species being absent from much of the Mediterranean, and is genetically distinct. Presumably the species once occurred in the Mediterranean for there to have been colonisation of the Black Sea.

Harbour porpoise populations form at least ten management units in western Europe, and an isolated population in the Black Sea (with small numbers in the neighbouring Aegean Sea). Several of these occur across Member State boundaries, and therefore the species is perhaps best listed under category S1 as having a clearly sustainable meta-population above Member State level. Supra-national management units can be listed under category S4. It is not generally considered to be a fully migratory species, although there is movement of individuals across national boundaries.

Step 1.3 Historical perspective: what happened to the species?
Although population surveys in Europe only started in the 1990s, several publications present evidence indicating widespread harbour porpoise declines during the middle of the last century (Verwey 1975, Teixeira 1979, Casinos & Vericad 1976, Evans 1980, 1990, 1992, Evans & Scanlan 1990, Smeenk 1987, Reijnders 1992, Tregenza 1992). Over that period, the species became rare in the southern North Sea, the English Channel, Bay of Biscay, and around the Iberian Peninsula. Since the 1990s, porpoises have returned in numbers to the southern North Sea and parts of the English Channel although they have become less abundant in the northernmost North Sea (Camphuysen 2004, Camphuysen & Peet 2006, Evans *et al*. 2003, 2008, Hammond *et al*. 2013).

There is uncertainty over the historical status of the species in the Mediterranean from which it is currently virtually absent, but during the latter half of the last century, numbers in the Black Sea were dramatically reduced by massive direct killing for the cetacean-processing industry that continued until 1983 (Smith 1982, IWC 2004, Birkun & Frantzis 2008). Large directed takes occurred during the years 1976-1983 before the ban on small cetacean hunting was declared in Turkey in 1983. Within that period, the total number of harbour porpoises killed was at least 163,000-211,000 (Birkun & Frantzis 2008). Illegal direct killing of unknown numbers continued in some parts of the Black Sea until 1991. Regionally extensive incidental mortality of porpoises in bottom-set gillnets is roughly estimated to have been in the thousands annually through the 1980s. The scale of this mortality almost certainly increased in the 1990s-2000s owing to the rapid expansion of illegal, unreported and unregulated fishing in the Black Sea (Birkun & Frantzis 2008).

Until 1983, unregulated hunting was the primary threat (IWC 1992, 2004). Very large numbers of harbour porpoises (as well as other cetaceans) were taken during the 20th century by all Black Sea countries for a variety of industrial uses (Kleinenberg 1956, Tomilin 1957). Although the total number killed is unknown, it may have been as many as four or five million for all species combined (e.g. see review in Smith 1982). It is widely accepted that all Black Sea cetacean populations, including Harbour Porpoises, were badly reduced by the directed fishery (IWC 1983, 1992, 2004). Catches of harbour porpoises were numerically fewer than those of common dolphins until 1964 when harbour porpoises became predominant (Danilevsky & Tyutyunnikov 1968, Smith 1982). Turkish catches of harbour porpoises in the early 1970s (see Berkes 1977) were thought to be at least as high as, and possibly much higher than, those estimated for 1976-1981 (34,000-44,000 per year according to IWC, 1983, assuming that harbour porpoises accounted for 80% of the total). At least since 1991, there has been no evidence of illegal directed takes although such takes had been reported before that time (IWC 1992).

At present, incidental mortality in bottom set gillnets is the most serious threat (*e.g.*, Birkun 2002a). Although all three Black Sea cetacean species are 'bycaught', the majority (95%) of recorded cetacean entanglements are of porpoises. Although absolute numbers of removals cannot be estimated from the available data., there are indications that the annual level of harbour porpoise bycatch may be in the thousands.

There have also been a number of mass mortality incidents of porpoises in the region. An explosion at a gas-drilling platform in the Azov Sea in August 1982 resulted in the deaths of over 2,000 porpoises (Birkun 2002b). Large-scale pelagic and small-scale coastal fisheries may affect Black Sea harbour porpoises indirectly by reducing their prey populations and degrading their habitat. Primarily, this relates to anchovies and sprats in the Black Sea and gobies in the Azov Sea. In particular, overfishing, eutrophication and the population explosion of an introduced predator, the ctenophore *Mnemiopsis leidyi*, led to a dramatic (8 to 12-fold) decline of sprat and anchovy abundance in the early 1990s (Prodanov *et al.* 1997). This reduced prey availability coincided with two mass mortality events (in 1989 and 1990) primarily affecting porpoises resulting from a severe nematode infection with bacterial complications (Birkun 2002c). This may have been exacerbated by immuno-suppression associated with PCB contamination Birkun & Frantzis 2008). Reported levels of DDTs and HCHs in Black Sea harbour porpoises at the time were higher than those in conspecifics elsewhere in the world (Tanabe *et al*. 1997). Chemical pollution is thus also a potential threat, particularly in the context of epizootics. Black Sea harbour porpoises are also affected in some years by ice entrapment in the Azov Sea

In the European Atlantic and North Sea, the main threat to harbour porpoise is thought to be by-catch, mainly from bottom set gill nets and pelagic trawls, with regions most affected including western Norway, central and southern North Sea, and Celtic Sea (Donovan & Bjørge 1995, Tregenza *et* al. 1997, Kaschner 2003, Vinther & Larsen 2004, Winship 2009, Bjørge *et al*. 2013, ICES 2015a, b). However, attacks by bottlenose dolphins (Ross & Wilson 1996, Jepson & Baker 1998), depletion of prey resources (Evans 1990, Reijnders 1992), noise disturbance (Carstensen *et al*. 2006, Brandt *et al.* 2011, Teilmann & Carstensen 2012), and infectious disease potentially arising from high contaminant levels (particularly of PCBs) (Bruhn *et* al. 1999, Kleivane *et* al 1999, Jepson *et al.* 2005, 2013, 2016, Hall *et al.* 2006) are also believed to pose threats. From post mortem examinations of 1,692 porpoises stranded in the UK between 1991 and 2010, 23% from infectious disease, 19% were thought to be the result of bottlenose dolphin attacks, 17% were diagnosed as by-catch, 15% died from starvation, and 4% live-stranded, (Deaville & Jepson 2011).

In inner Danish waters, harbour porpoises are thought to have been directly exploited since the Stone Age (Möhl 1970), with written records dating from the fourteenth century (Kinze 1995). The overall average annual take was thought to have been c. 1,000 animals with a known minimum total of 59,028 for the years 1819-92 (Kinze 1995). In the 1880s, the annual catch level increased to c. 2,000 of a grand total of c. 3,000 animals in all Danish waters. The actual catch may have been sustainable for several centuries until about 1870 (Kinze 1995). In the Lille Belt area, the take occurred annually in winter and spring, and consisted of migrating animals with summer ranges thought to extend into the Baltic proper. The population in the Baltic proper is currently estimated at only c. 600 animals (SAMBAH 2016), attributed to a marked decline in the second half of the twentieth century (Kinze

1995). The hunt ceased in the first half of the twentieth century although it is not clear whether the decline had already begun by then. The historic range of the harbour porpoise extended into the north-eastern parts of the Baltic Sea but during the second half of the twentieth century, when numbers declined, the distribution range narrowed, and now the species is largely confined to the western end (Koschinski 2002, SAMBAH 2016). It has often been concluded that porpoises escaped from ice cover in the eastern Baltic Sea in the winter and re-colonised the Baltic Proper in spring (Koschinski 2002). However, more recent observations, as described above, indicate that migration behaviour is much more complex and diffuse (Verfuß *et al.* 2007, Sveegaard *et al*. 2011, Gallus *et al*. 2012, Benke *et al.* 2014).

Concern has been raised about the long-term viability of the harbour porpoise population in the Baltic Proper. Bycatch at possibly unsustainable levels, contaminants, overfishing of prey species and disturbance have all been identified as potential threats (Koschinski 2002, SAMBAH 2016). Environmental contaminants most likely affect the long-term viability of Baltic Sea harbour porpoise stocks and might have been a major cause for the apparent decline between the 1940s and the 1970s (Teilmann & Lowry 1996, Koschinski 2002). Since then, concentrations of PCBs and other organochlorine contaminants have declined. To date, the most important threat to Baltic Sea harbour porpoises is bycatch in salmon drift nets and bottom-set gillnets (for cod and other demersal species) (Lindroth 1962, Skóra *et al.* 1988, Christensen 1991, Skóra 1991, Berggren 1994, Kock & Benke 1996), although noise pollution has the potential to increasingly become a threat due to the development of new activities in the Baltic Sea (e.g. offshore wind farms, fast ferries - Koschinski 2002, Teilmann & Carstensen 2012).

Climate change may also affect harbour porpoises either by altering human activities and thus pressures or by affecting the stock sizes and distribution of their prey (Evans & Bjørge, 2013). There is some evidence that changes in the stock sizes of initially herring and subsequently sand eel may have resulted in shifts in the distribution of porpoises in the North Sea and Danish waters (Evans 1990, Reijnders 1992, Hammond *et al*. 2013). In the past, porpoises in the Baltic Proper are reported to have experienced periodic catastrophic mortality from severe winter ice conditions (Lindroth 1962, Berggren 1994, Teilmann & Lowry 1996).

Step 1.4 Analysis of distribution and trends.

Population estimates do not exist for the entire North Atlantic range of the harbour porpoise, or even for the European range. However, the widest scale surveys were SCANS undertaken in 1994, followed by SCANS-II in 2005, and SCANS-III in 2016. From line transect SCANS surveys in July 1994 (Hammond *et al.* 2002), an overall population estimate of 341,000 porpoises (CV=0.14; 95% CI: 260,000-449,000) was made, with the following regional estimates: North Sea (c. 250,000), NW Scotland (c. 18,000), Baltic region (36,600 in Kattegat / Skagerrak / Belt Seas / Western Baltic Sea), Channel (0), and Celtic Shelf (36,300). Only a small portion of the Baltic Proper was surveyed.

A repeat survey in July 2005 (SCANS-II), covering a wider area (continental shelf seas from SW Norway, south to Atlantic Portugal), gave an estimate of 375,358 (CV=0.20; 95% CI: 256,300-549,700) (Hammond *et al*. 2013), with regional estimates: North Sea including Shetland & Orkney (c. 191,500), Baltic (19,100 in E. Skagerrak / Kattegat / Belt Seas / Western Baltic Sea), Channel (40.900), Celtic Shelf (72,400), Irish Sea (15,200), Atlantic Ireland & Scotland (33,800) and the Iberian Peninsula & SW France (2,400).

Comparing the two surveys, although the overall number estimated for the North Sea, Channel and Celtic Sea was comparable (341,000 in 1994, and 305,000 in 2005), numbers in the northern North Sea and Danish waters had declined from 239,000 to 120,000, whereas in the central and southern North Sea, Channel and Celtic Shelf, they had increased from 102,000 to 215,000. This is thought to represent a southward range shift rather than actual changes in population size (Winship 2009, Hammond *et al*. 2013), at least for the month of July. In Norwegian waters, estimates of 11,000 porpoises (95% CI: 4,790-25,200) for the Barents Sea and Norwegian waters north of 66^{o}N, and 82,600 (95% CI: 52,100-131,000) for southern Norway and the northern North Sea, were made during July 1989 (Bjørge & Øien 1995). There have been no abundance estimates for porpoises covering Norwegian waters since then.

A third SCANS (SCANS-III) survey was undertaken in July 2016. Taking equivalent areas between the three surveys, they revealed no significant change, with 407,177 (CV 0.18) in 1994, 519,864 (CV 0.21) in 2005/07, and 466,569 (CV 0.15) in 2016 (P.S. Hammond *pers. comm.*). The southwards shift in abundance of porpoises in the North Sea that was observed between 1994 and 2005 has persisted through to 2016.

The number of porpoises in the Baltic Proper has recently been estimated at 497 individuals (95% CI 80-1091), using sonar click detectors (C-PODs) deployed at a total of 304 locations spread around the region between 2011 and 2013 (SAMBAH 2016).

<u>Conclusions</u>
The harbour porpoise is thought to have been much more widespread and abundant in the shelf seas of Europe in previous centuries. Although quantitative information is largely lacking, the species appears to have experienced widespread declines in the middle of the twentieth century. Since the mid 1990s, populations of harbour porpoise in the Atlantic, North Sea and Danish Belt Seas may have remained stable. On the other hand, although direct hunting in those regions has ceased, the genetically distinct porpoise populations in the Baltic Proper and the Black Sea continue to face serious pressures from fisheries bycatch and a range of other human activities, with little signs of recovery. Current threats to populations throughout Europe include fisheries bycatch, depletion of prey resources, pollution and noise disturbance. If those human pressures are reduced, there is scope for recovery of depleted populations.

Ideally, population trends should be considered separately for each of the Management Units that have been recognised. Up to now, that has not been done although the three large-scale synoptic surveys conducted in northern Europe at decadal intervals have enabled abundance estimates to be made by ICES sub-area, and some of these more or less coincide with MUs. However, the confidence limits tend to be too wide at a regional scale to robustly determine trends. These surveys started in the mid-1990s after the Habitats Directive came into force. Thus, there are no data prior to human impacts in these areas, so it is not possible to set a historical baseline. Although the historical abundance and distribution is unknown, there is good evidence that the species was once more widely distributed in western Europe, and more abundant in the Baltic Proper and the Black Sea. The results of those surveys at a larger scale show no significant trend over the last twenty years.

Step 2.1 FRP assessment
Scarcely any quantitative information exists of population numbers for bottlenose dolphin before the Habitats Directive came into force in 1992, despite the fact that this species is the best known of all cetaceans. Nevertheless, there is evidence that the species was more abundant in the Baltic Proper and Black Sea in historical times, and that in North-west Europe (particularly the North Sea and English Channel), it experienced widespread declines in the middle of the twentieth century from which it may then have partially recovered.

The indication therefore is that FRP for at least some of the management/assessment units should be above the current values. This is clearest for the Black Sea where habitat degradation and fisheries conflicts as well as past human exploitation have almost certainly depressed populations well below their natural carrying capacity. The same applies to the Baltic Proper which has been exposed to similar pressures. Thus, if conditions are improved, one might expect populations to increase, and to recolonise areas (e.g. eastern Mediterranean and eastern Baltic respectively), where the species is currently very rare. The recent shifts in centres of abundance observed in the North Sea very probably relate mainly to changes in food availability. Whether or not this affects the overall population size for different management/assessment units is unclear. Further south, there is a hiatus in the Bay of Biscay where the species is uncommon. This may relate primarily to the bathymetry of most of the

Bay which is at greater depths to that of the favoured foraging habitat of porpoises. Recovery in this area may therefore be more constrained, although genetic studies indicate there is mixing of populations taking place along the French Atlantic coasts from both north and south.

As mentioned for bottlenose dolphin, whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

The order of magnitude of FRP is difficult to determine but is likely to be in the range of one to five million animals. This is for the species throughout Europe and including the Black Sea. For individual MUs, it will be much less.

Conclusions
FRP>>CV (Baltic MUs in MBAL, Black Sea MUs in MBLS)
FRP>CV (Atlantic MUs in MATL)

Step 2.2 FRR assessment
Harbour porpoises are thought to have occurred throughout the shelf seas of Europe. Nowadays it is largely absent from the Mediterranean Sea and the eastern part of the Baltic Proper. The current range could therefore extend to include those areas if suitable conditions prevailed.

Conclusions
FRR >> CV (Baltic MUs in MBAL, and Mediterranean MUs in MMED)
FRR > CV (Black Sea MUs in MBLS, Atlantic MUs in MATL)

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1.4 Humpback whale (*Megaptera novaeangliae*) in the European Atlantic

Step 1.1 Biology of the species

The humpback whale (*Megaptera novaeangliae*) is a large rorqual from the family Balaenopteridae that includes blue, fin, sei and minke whale. The species has no strongly organised social structure. It is typically found singly or in small unstable groups, although large feeding aggregations can occur in summer, and large competitive groups of males can form around females in breeding areas. Group size is often associated with the size of the exploited prey patch, although due to the severe historical overexploitation of the species, aggregations rarely number more than ten individuals. Breeding is strongly seasonal, with most births in low latitudes in winter (peak January-March). The age at which humpbacks become sexually mature varies by population, from 4 to >10 years for both sexes. Once mature, females give birth to a single calf typically every 2-3 years, with occasional annual calving. Longevity is at least 48 years, and possibly much longer. The generation length has been estimated at 14.5 years (21.5 years in populations at a stable state). The generalist diet includes both shoaling fish, notably herring, sprat, sandeel, mackerel and, in polar regions, capelin, and plankton (mainly euphausiids). More detailed accounts of the biology of the species can be found in Clapham (1996), Claphan & Mead (1999), Clapham & Evans (2008), and Clapham (2009).

Step 1.2 Spatial scale of functioning

The humpback whale has a worldwide distribution in all seas, occurring even occasionally to the ice edge. It is a highly migratory species, feeding in summer in high latitudes, and mating and calving in winter in tropical waters, although a few overwinter on the feeding grounds. The species shows strong individual fidelity to feeding areas; in the North Atlantic, these include the Gulf of Maine, Gulf of St Lawrence, Newfoundland/Labrador, Greenland, Iceland and Norway. Matching of photographically and genetically identified individuals indicates that the eastern North Atlantic population migrates primarily to the West Indies (Martin *et al*. 1984, Stevick *et al.* 1998, 2003, 2006), although some animals winter in the Cape Verde Islands (Reiner *et al*. 1996, Hazevout & Wenzel 2000, Jann *et al*. 2003); genetic analysis suggests a third, unknown, breeding area. Despite fidelity to specific feeding grounds, however, whales from all North Atlantic areas appear to mix spatially and genetically in the West Indies in winter.

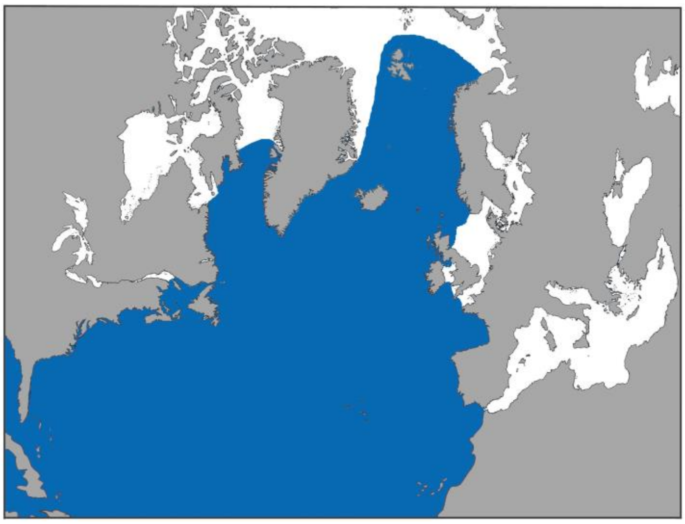

Figure 1.4.1 *North Atlantic distribution of humpback whale (depicting those areas where the species is thought to regularly occur)*

The species is more common on the west side, spending the summer in Baffin Bay and along the New England coast. In the eastern North Atlantic it occurs around Iceland, Norway, the British Isles and Ireland. North Atlantic humpbacks winter in the West Indies with small numbers around the Cape Verde Islands off North-west Africa (Fig. 1.4.1). The species is only a casual visitor to the Mediterranean and the Baltic Seas (Notarbartolo di Sciara & Birkun 2010, Kinze *et al.* 2010).

In European shelf waters, humpback whales occur mainly between May and September, and this was the period when catches from whaling activities were highest (Thompson 1928, Brown 1976, Clapham & Evans 2008). However, particularly from the British Isles and Ireland south to the Iberian Peninsula, sightings may occur at any time of year, including November to March, as indicated also from acoustic detections with SOSUS hydrophone arrays offshore in the North Atlantic at these latitudes (Clark & Charif 1998, Charif *et al*. 2001). Sightings in Ireland, occurring mainly along the south coast, increase through the summer to peak in September to December, rapidly declining between January and May (Berrow *et al*. 2010).

Six distinct feeding aggregations have been identified in the North Atlantic: Gulf of Maine; Gulf of St Lawrence; Newfoundland/Labrador; West Greenland; Iceland; and North Norway (including Bear Island and Jan Mayen) (Reilly *et al*. 2008). Genetic and photo-ID data indicate that the six feeding aggregations represent relatively discrete subpopulations, fidelity to which is determined matrilineally. However, because whales from different feeding grounds all mix in a common breeding area in the West Indies, there is male-mediated nuclear gene flow between the subpopulations.

<u>Conclusions</u>
Current knowledge indicates that the humpback whale in the Atlantic, which is its main range in Europe, should be listed under category MNR4, a non-reproductive population of a migratory species with individuals with large home ranges and showing large cyclic, directed movements, although it is possible that a segment of the North Atlantic population belongs in category MNR1, as a largely non-reproductive population of a (partial) migratory species with clearly sustainable size and extent above Member State. The same population can occur in different Member States in time.

Step 1.3 Historical perspective: what happened to the species?
Humpbacks were heavily exploited in the past by pre-modern whaling in their breeding grounds in both the West Indies and the Cape Verde islands, and by modern whaling in their feeding grounds, especially off Iceland and off Norway in the late 19th and early 20th centuries (Reilly et al. 2008). Catches of pre-modern whaling are estimated primarily from trade records. Catches of early modern whaling also need to be estimated, because most of the catch records were not divided by species. Catches in the West Indies (including Bermuda) are documented from 1664 to the present day, but the main period was 1826-1928, during which about 8,600 whales were estimated to have been killed. Whaling in the Cape Verde Islands occurred primarily during 1850-1912 with a total estimated kill of about 3,000 animals. An estimated 3,200 were taken from Iceland and 2,000 from northern Norway during 1880-1916 (Reilly et al. 2008). About 1,500 humpback whales are reported to have been killed in the North Atlantic since 1916, from a variety of areas including the British Isles, Faroe Islands, Norway, Iceland, Greenland, and eastern Canada, as well as Norwegian pelagic catches (Reilly et al. 2007). However, humpbacks are no longer exploited in Europe.

There remains much debate over what the pre-exploitation population sizes were for humpback whale (and other baleen whales), due to inaccuracies in the catch record, uncertainties surrounding genetic estimates, and/or differences in time scales applied to the estimates (Roman & Palumbi 2003, Holt & Mitchell 2004, Punt et al. 2006, Alter & Palumbi 2009, Smith & Reeves 2010, Ruegg et al. 2013). Nevertheless, they generally indicate marked reductions in population sizes for most of the baleen whale species since commercial whaling started. Based upon mtDNA analyses of genetic diversity, for example, the pre-exploitation population size of North Atlantic humpback whales has been estimated at 240,000 compared with a “current” estimate of 9,300-12,100 (Roman & Palumbi 2003). Ruegg et al. (2013), estimated the long-term population size of c. 112,000 individuals, based on nuclear gene diversity, which although lower than the mtDNA estimate, is still 2-3x higher than estimates based upon catch data. Commercial whaling of humpback whales in the North Atlantic was banned by the IWC in 1955, by which time it was estimated that this population had been reduced to 1,000 individuals (Mitchell & Reeves 1983, Katona & Beard 1990). So, whatever the historical population size, humpbacks in the North Atlantic do appear to be steadily increasing though probably still nowhere near pre-exploitation levels.

The other known causes of mortality are bycatch, mainly through entanglement in ghost netting or creel lines (Northridge et al. 2010; Ryan et al., 2016) and ship strikes (Evans et al. 2011), although the species may be affected by noise disturbance from seismic surveys (Risch et al. 2012, Cerchio et al., 2014). Prey depletion through overfishing or the effects of climate change may also have an influence on humpback whale abundance and distribution (Evans & Bjørge 2013).

Step 1.4 Analysis of distribution and trends.

Overall, the North Atlantic population has recovered well from exploitation, estimated at somewhere between 9,400-16,400 in 1992, with the great majority occurring in the west-central part (Smith *et al*. 1999, Stevick *et al*. 2003). NASS surveys around Iceland in the central North Atlantic gave an abundance estimate of 10, 521 (95% CI 3,700-24,600) in 1995, and 14,662 (95% CI 9,400-29,900) in 2001, mainly to the east and north of Iceland but also to the west (Paxton *et al.* 2009). In the Barents and Norwegian Seas, the Norwegian survey estimate was 1,059 (CV=0.25) in 1995, and 1,450 (CV=0.29) in 1996-2001 (Øien 2009). Recent survey estimates suggest that the North Atlantic population may be approaching 20,000 animals (Smith and Pike 2009). During SCANS-II and CODA surveys, numbers observed in west European waters were too low to derive abundance estimates.

Sightings from around the British Isles and Ireland have increased markedly since the early 1980s; occurring in three main areas – The Northern Isles south to eastern England; The northern Irish Sea north to West Scotland; and the Celtic Sea between Southern Ireland, Southwest Wales and Southwest England, with a few sightings and strandings also in the southern North Sea (Evans *et al*. 2003, Camphuysen & Peet 2006, Clapham & Evans 2008].

<u>Conclusions</u>

The humpback whale was intensively hunted in the European Atlantic during the nineteenth and first half of the twentieth century. This is thought to have caused major reductions in population size. Whaling has now largely ceased in the North Atlantic. Since the moratorium on commercial whaling in the mid 1950s, the overall population appears to be steadily increasing.

Other human pressures such as bycatch, ship strike and noise disturbance exist but may not be of a magnitude to have population level effects. Changes in prey abundance and distribution as a result of overfishing or the effects of climate change may also have an impact.

Step 2.1 FRP assessment

As a result of many years of intense human exploitation, population levels for humpback whales in the European Atlantic will have been much lower at the time when the Habitats Directive came into force than historically before exploitation, and so Favourable Reference Population size should exceed Current Value, although by what magnitude is very difficult to say with confidence. Genetic analyses at the mtDNA control region suggests that FRP could be two or three times the Current Value, although this is based partly on what the generation length is set at (Roman & Palumbi 2003).

Whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

Conclusions
The order of magnitude of FRP in the North Atlantic is likely to have been in the range of 100,000-250,000 humpback whales, based upon genetic evidence. The bulk of the North Atlantic population breeds in the West Indies, some of which then migrate into northern European waters around Iceland and northern Norway. The declines that have occurred historically could continue to be reversed if conditions allow.

FRP >> CV (European Atlantic, MATL and MMAC)

Step 2.2 FRR assessment
The range of the humpback whale in Europe does not appear to have changed much from what it was during the period of commercial exploitation, although by the latter half of the twentieth century, humpbacks had become very scarce in the vicinity of the British Isles (Evans *et al*. 2003). There is no historical evidence to suggest the species was ever common in the Mediterranean or the Baltic. That FRR > CV relates to it being regular in its former range in areas where it is currently only an occasional visitor.

Conclusions
FRR > CV (European Atlantic, MATL and MMAC).

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1.5 Minke whale (*Balaenoptera acutorostrata*) in the European Atlantic

Step 1.1 Biology of the species
The minke whale (*Balaenoptera acutorostrata*) is the smallest of rorqual whales of the family Balaenopteridae occurring in the North Atlantic. It is usually seen singly or in groups of 2-3 individuals although it may aggregate into larger groups of tens or even hundreds of individuals when in the vicinity of prey concentrations. Minke whales frequently approach vessels and may bow- or stern-ride them. Evidence from recognisable individuals indicates some seasonal site fidelity over a small gepgraphic range. Differential migration by sex and age may lead to segregation by sex and breeding condition. Breeding is diffusely seasonal, possibly occurring mainly in wintertime. Females are thought to reach sexual maturity around 6 years, and males at 7 years of age. Once sexually mature, females give birth to a single calf annually or every 2 years. Longevity is 40-50 years although the maximum age recorded is 57.5 years. Generation length has been estimated at 13.0 years (22.1 years in populations at a stable state). The diet comprises mainly shoaling meso- and benthopelagic fish, and includes herring, sandeel, sprat, cod, haddock, saithe, whiting, mackerel, blue whiting and pouts, as well as small cephalopods. In polar regions, capelin and plankton (copepods, euphausiids) may predominate. More detailed accounts of the biology of the species can be found in Stewart & Leatherwood (1985), Horwood (1990), Anderwald *et al*. (2008), and Perrin & Brownell (2009).

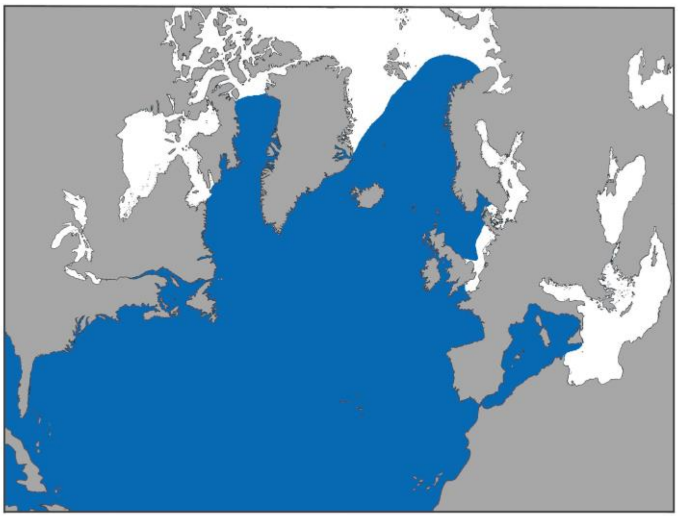

Figure 1.5.1 *North Atlantic Distribution of Minke Whale (depicting those areas where the species is thought to regularly occur)*

Step 1.2 Spatial scale of functioning
The species has a cosmopolitan distribution from the tropics to the ice edge in both hemispheres, though more uncommon in equatorial waters (Fig. 1.5.1). It is widespread along the Atlantic seaboard of Europe from Norway south to the southern tip of Portugal, as well as in the North Sea, although abundance is greatest in the north (Reid *et al*. 2003, CODA 2009, Hammond *et al*. 2013). Around the British Isles and Ireland, highest numbers occur off the north and west coasts of Scotland and the Hebrides, the west and south coasts of Ireland, central part of the Irish Sea including the Celtic Deep, and in the northern and central North Sea including around the Dogger Bank; it is rare in the southernmost North Sea and eastern half of the English Channel (Evans *et al*. 2003, Reid *et al*. 2003, Camphuysen & Peet 2006, Robinson *et al*. 2009, De Boer 2010, Berrow *et al*. 2010, Anderwald *et al*. 2011, Baines & Evans 2012, Hammond *et al*. 2013). In the western English Channel south to southwest Portugal, it is present but uncommon out to the edge of the continental shelf, but is largely absent from the deeper parts of the Bay of Biscay (Evans *et al*. 2003, Reid *et al*. 2003, CODA 2009, Hammond *et al*. 2013). The species is a casual visitor to Inner Danish waters, the Baltic and

Mediterranean (mainly the western part) (Kinze *et al*. 2010, Notarbartolo di Sciara & Birkun 2010). Most of the occasional strandings and sightings in the Mediterranean occurred in the Algero-Provençal and Tyrrhenian subregions, although rare occurrences are known from the Levantine Sea off Israel.

Minke whales from high latitudes are thought to migrate southwards to winter in lower latitudes (Risch *et al*. 2014); at mid-latitudes, however, such as around the British Isles and Ireland, at least some have been recorded in every month of the year (Anderwald & Evans 2007).

There is some genetic evidence for two sympatric stocks existing in the North Atlantic, with overlapping ranges, but otherwise no evidence of population structure (Anderwald *et al*. 2011). The implication is that minke whales range extensively across the North Atlantic seasonally, but segregate to some extent on at least two breeding grounds (as yet unidentified).

The population category should be largely the same as humpback whale (MNR4) except that reproduction most likely can occur within the region, and movements may not necessarily be directional – they may simply disperse for the most part off the continental shelf in winter. Although there is some summer site fidelity, annual home ranges almost certainly exceed MS level.

Conclusions
Current knowledge indicates that the minke whale in the Atlantic and North Sea, which is its main range in Europe, should be listed either under category MR4, a population of a migratory species with individuals with large home range and showing large cyclic, directed movements. The same population can occur in different Member States in time. It is not known whether migratory behaviour results in non-reproductive populations in Europe.

Step 1.3 Historical perspective: what happened to the species?
Whaling for minke whales in the North Atlantic dates back to at least the Middle Ages when a fishery existed near Bergen, Norway (Collett 1912). However, it was not until the 1920's that it became a target of commercial whaling when stocks of the larger baleen whales like blue, fin and sei whale had become so depleted that it was no longer economic to target them. Exploitation was greatest between the 1940s and 1980s, with recorded catches totalling about 140,000 (Mitchell 1975, Reilly *et al.* 2008). The largest catches were by Norwegian "small-type" whalers who have taken about 120,000 since 1948, mainly in the Northeast Atlantic. Annual catches peaked at over 4,000 in the late 1950s, declining to about 2,000 annually in the early 1980s. Whaling in Iceland began in 1914 and between then and 1980, approximately 3,362 minke whales were taken (Sigurjonsson 1980). Catches were phased out from 1984 to 1987. Commercial minke whaling resumed in 1993 at a lower level and continues to the present.

Calculations of pre-exploitation population sizes have been fraught with difficulties. The often widely different estimates obtained may be due to: inaccuracies in the catch record, uncertainties surrounding genetic estimates, and/or differences in time scales applied to the estimates (Roman & Palumbi 2003, Holt & Mitchell 2004, Punt *et al*. 2006, Alter & Palumbi 2009, Smith & Reeves 2010, Ruegg *et al*. 2013). However, they generally indicate marked reductions in population sizes for most of the baleen whale species since commercial whaling started. Based upon mtDNA analyses of genetic diversity, for example, the pre-exploitation population size of North Atlantic minke whales has been estimated at 265,000 compared with a "current" estimate of 149,000 (Roman & Palumbi 2003). Since the moratorium on commercial whaling was imposed by the IWC in 1986, whaling has largely ceased and populations of a number of species including minke whale appear to be recovering. The most recent estimate for the entire North Atlantic was 182,000 (Reilly *et al*. 2008).

The minke whale is still exploited in Atlantic Europe by Norway, under objection of the IWC, resulting in c. 24,300 animals taken since 1978 (averaging 200-600 per year). However, there is no indication of a decline in numbers of minke whales in NW European seas since the mid 1980s, and indeed there may have been an increase in some areas (Schweder *et al*. 1997, Evans *et al*. 2003, Skaug *et al*. 2004, Bøthun *et al*. 2009, Hammond *et al*. 2013, Paxton *et al.* 2016).

The other known causes of mortality are bycatch, mainly through entanglement in ghost netting or creel lines (Northridge *et al*. 2010) and ship strikes (Evans *et al*. 2011), although the species may be affected by noise disturbance from seismic surveys and the use of mid-frequency active sonar (Kvadsheim *et al*. 2017). Prey depletion through overfishing or the effects of climate change may also have an influence on minke whale abundance and distribution (Evans & Bjørge 2013).

Step 1.4 Analysis of distribution and trends.
Population estimates for minke whales in the Atlantic from southern Norway southwards including the North Sea (i.e. EU range states) are from the SCANS, SCANS-II and CODA, and SCANS-III surveys. In July 1994, the SCANS survey of the North Sea, English Channel and Celtic Sea estimated 8,450 individuals (95% CI: 5,000-13,500) (Hammond *et al*., 2002). A more extensive line transect survey (SCANS-II) over the North West European continental shelf in July 2005 gave an overall estimate of 18,958 (CV=0.35; 95% CI: 9,800-36,700) (Hammond *et al*. 2013). And the offshore CODA survey in 2007 yielded a population estimate of 6,765 (CV=0.99; 95% CI: 1,300-34,200) (CODA 2009). This latter estimate has very wide confidence intervals and was uncorrected for animals missed along the track-line, and is therefore negatively biased. For the equivalent area surveyed in 1994 and 2005, there was no evidence for a significant change in numbers (Hammond *et al*. 2013). The SCANS-III survey undertaken in July 2016 yielded an abundance estimate of 13,101 (CV=0.35; 95% CI: 7,050-26,700) (Hammond *et al*. 2017). This estimate applies to West Norwegian coastal waters, North Sea (including English Channel), West Scotland, Irish Sea, Celtic Sea, Bay of Biscay, and Iberian Peninsula, but excludes west and south of Ireland and offshore Portugal.

A population estimate for the entire North Eastern North Atlantic (based upon data from 2008-2013) gave 90,000 individuals (95% CI: 60,000-130,000) (IWC website: <www.iwc.int>), with an additional 50,000 (95% CI: 30,000-85,000) in the central North Atlantic (2005-2007) (see also Lockyer & Pike 2009). Previously, the stock seasonally inhabiting the Norwegian and Barents Seas was estimated at 86,700 individuals (95% CI: 61,000-117,000) (Schweder *et al*. 1997). Assessing minke whale numbers is difficult and controversial, since the species is inconspicuous at sea, and often reacts to survey vessels. Nevertheless, the current population appears to be in a relatively healthy state.

Conclusions
The minke whale was intensively hunted in the European Atlantic throughout much of the twentieth century. This is thought to have caused major reductions in population size by the 1980s. Whaling continues but at a much lower level, and may no longer be limiting population growth. Other human pressures such as bycatch, ship strike and noise disturbance exist but may not be of a magnitude to have population level effects. Changes in prey abundance and distribution as a result of overfishing or the effects of climate change may also have an impact.

Since the moratorium on commercial whaling in the mid 1980s, numbers taken (under objection of IWC) have been in the order of 200-600 per annum, and the overall population may now be stable or increasing.

Step 2.1 FRP assessment
As a result of many years of intense human exploitation, population levels for minke whales in the European Atlantic will have been much lower at the time when the Habitats Directive came into force than historically, and so Favourable Reference Population size should exceed Current Value, although by what magnitude remains difficult to say with confidence. Genetic analyses at the mtDNA control region suggests that FRP could be as much as twice the Current Value (Roman & Palumbi 2003).

Whole genomic analysis using restriction site associated DNA sequencing is recommended for insights into the extent to which present day populations have experienced contractions in size and loss of genetic diversity. Genetic analysis enables one to estimate the effective population size (Ne) for management units prior to major human impacts.

The order of magnitude of FRP in the North Atlantic is likely to be in the range of 200,000-300,000 minke whales, of which maybe 60% occur in the European Atlantic. The species has a more or less continuous distribution which in summer is concentrated in the northern North Atlantic (where females may constitute c. 70% or more) (Horwood 1987, Anderwald *et al*. 2011). The declines that have occurred historically could be reversed if conditions allowed.

Conclusions FRP > CV (European Atlantic, MATL and MMAC)

Step 2.2 FRR assessment

The range of the minke whale in Europe does not appear to have changed much from what it was during the period of commercial exploitation, the whaling grounds being the same areas as those occupied today (Mitchell 1975, Horwood 1987). There is no historical evidence to suggest the species was ever common in the Mediterranean or the Baltic.

Conclusions FRR = CV (European Atlantic, MATL and MMAC)

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