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TRENDS IN AQUATIC ECOLOGY

Review Paper

Marine mammals and good environmental status: science, policy and society; challenges and opportunities

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Abstract The Marine Strategy Framework Directive has become the key instrument for marine conservation in European seas. We review its implementation, focusing on cetacean biodiversity, using the examples of Spain and the Regional Seas Convention, OSPAR. The MSFD has been widely criticised for legal vagueness, lack of coordination, uncertainty about funding, and poor governance; its future role within EU Integrated Maritime Policy remains unclear. Nevertheless, the first stages of the process have run broadly to schedule: current status, environmental objectives and indicators have been described and the design of monitoring programmes is in progress, drawing on experience with other environmental legislation. The MSFD is now entering its critical phase, with lack of funding for monitoring, limited scope for management interventions, and uncertainty about how conservation objectives will be reconciled with the needs of other marine and maritime sectors, being among the main concerns. Clarity in governance, about the roles of the EU, Member States, Regional Seas Conventions and stakeholders, is needed to ensure success. However, even if (as seems likely) good environmental status cannot be achieved by 2020, significant steps will have been taken to place environmental sustainability centre-stage in the development of Integrated Maritime Policy for EU seas.

Keywords MSFD · Cetaceans · GES · OSPAR · Spain

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Introduction

The EU's Marine Strategy Framework Directive (MSFD, 2008/56/EC) has been described as the key environmental instrument of European Union (EU) maritime policy, one designed to formalise an ecosystem-based approach to marine environmental management (De Santo, 2010; Bellas, 2014).

The MSFD was adopted in June 2008 and transposed into National Legislation by EU Member States in 2010. The overall aim of the MSFD is to enable sustainable use of marine goods and services by effectively managing human activities and pressures through an ecosystem-based approach. To achieve this aim, it sets the requirement that Member States must achieve (or maintain) good environmental status (GES) across all European waters by 2020. GES is defined as being reached when "*the overall state of the environment in marine waters provides ecologically diverse and dynamic oceans and seas which are healthy and productive*".

As part of the requirements of the MSFD (see Fig. 1), Member States (who are ultimately responsible for the implementation of the Directive in their national waters) were required to develop a marine strategy (Article 5), to make an initial assessment of their marine waters including an analysis of the essential features and characteristics and the predominant pressures and impacts (Article 8), to determine what GES means for their waters (Article 9) and to establish a list of environmental targets (Article 10). This first phase of the implementation was due to finish in 2012, although some delays occurred. The next phase, which is currently taking place, is the establishment (and implementation) of monitoring programs for the ongoing assessment of the status of marine waters (Article 11) and, by 2016, Member States are required to have put in place a programme of measures specifically designed to achieve (or maintain) GES in their waters (Article 13). This programme of measures should be developed in 2015.

The Directive sets out eleven qualitative descriptors of GES (Box 1) and specifically refers to conserving biodiversity, with Descriptor 1 stating that GES will be achieved when "*Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climate conditions*".

Assessment and reporting of "biodiversity" is a huge undertaking and guidance has been provided which, in the case of widely dispersed or highly mobile species (such as birds, mammals, reptiles, fish and cephalopods), suggests focusing at the level of "functional groups" (Cochrane et al., 2010). Marine mammals are included as one of these functional groups (defined as "an ecologically relevant set of species", see glossaries in Supplementary Material Annex 1 and 2), with four ecotypes described within the group as distinct "biodiversity components", namely baleen whales, toothed whales, seals and ice-associated mammals. These biodiversity components were selected to be used for the initial evaluation of environmental status, for the development of indicators to summarize available information, and to measure progress towards the achievement of GES (Cochrane et al., 2010).

Although the definition of GES ultimately lies with the Member States, the EU has published a Commission Decision (2010/477/EU) *on Criteria and Methodological Standards on Good Environmental Status of Marine Waters* to help guide Member States to develop a set of criteria to achieve GES (and

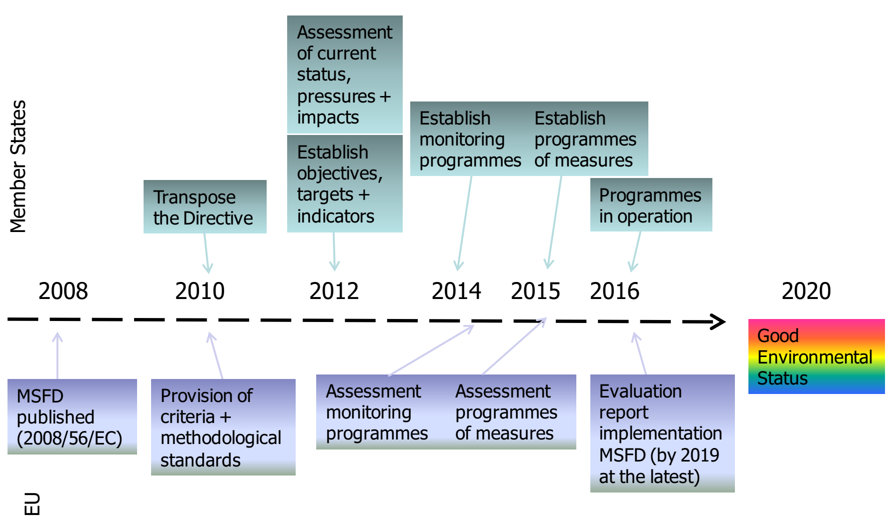

Fig. 1 Schematic representation of the road map for the implementation of the Marine Strategy Framework Directive, showing actions of Member States and the European Union

Box 1 Marine Strategy Framework Directive (2008/56/EC) descriptors of good environmental status

Descriptor	Relevance of marine mammals
(1) *Biological diversity* is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions	Marine mammals are one of the functional groups for biodiversity monitoring that are included by most Member States, covering their abundance, range, and population parameters, as well as impacts of threats: notably fishery bycatch but potentially also pollution, underwater noise, ship strikes, etc. Monitoring mainly comprises sightings surveys, strandings monitoring and onboard monitoring of bycatch
(2) *Non-indigenous species* introduced by human activities are at levels that do not adversely alter the ecosystems	n/a
(3) Populations of all *commercially exploited fish and shellfish* are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock	n/a
(4) All elements of the *marine food webs*, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity	As all European marine mammals are upper level predators, their abundances, ranges and population parameters (as covered under D1) are also all relevant to this descriptor
(5) *Human-induced eutrophication* is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters	n/a
(6) *Sea-floor integrity* is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected	n/a
(7) Permanent alteration of *hydrographical conditions* does not adversely affect marine ecosystems	n/a
(8) *Concentrations of contaminants* are at levels not giving rise to pollution effects	Monitoring for D8 typically includes fish, shellfish and sediments. However, as marine mammals bioaccumulate persistent organic pollutants such as PCBs, especially in their blubber (e.g. Aguilar et al., 1999), they could provide a useful indicator for these compounds. Samples can be collected as part of strandings monitoring
(9) *Contaminants in fish and other seafood* for human consumption do not exceed levels established by Community legislation or other relevant standards	n/a in the EU but if equivalent monitoring were undertaken in Norway, Iceland, Faroe and Greenland, monitoring would logically include marine mammal species harvested for human consumption
(10) Properties and quantities of *marine litter* do not cause harm to the coastal and marine environment	Ingestion of plastics can cause mortalities in marine mammals (e.g. Laist, 1987) as can entanglement in discarded fishing gear. Such mortalities can be detected through strandings monitoring
(11) Introduction of *energy, including underwater noise*, is at levels that do not adversely affect the marine environment	Underwater noise can have a range of effects on marine mammals, from disturbance to hearing loss and mortality. Particular concerns relate to seismic surveys and navel sonar (Parsons et al., 2009), the letter being associated with several mass strandings of beaked whales (e.g. Simmonds & López-Jurado, 1991; Fernández et al., 2013). Some mortalities are detected by strandings monitoring; specific monitoring. Marine mammal observers routinely participate in seismic surveys

environmental targets) that ultimately ensures comparability and facilitates coordination between Member States. This Commission Decision also provides methodological standards to facilitate a coherent process at EU level.

In the case of Descriptor 1, Member States were required to report their assessment at three separate ecological levels: ecosystems, habitats and species. For the species level, the Commission Decision supplied criteria for use in the assessment related to species distribution, population size and population condition. However, different Member States selected different species, based on species presence in their waters and information available, and taking into account obligations under existing legislation (e.g. Habitats Directive, 92/43/CEE).

The Directivés ultimate focus is on the whole marine environment, as part of the process to align EU legislation with the aim of the UN (and many other international and national organisations) to achieve sustainable development. Sustainability is frequently described as resting on three pillars, social, economic and environmental, and the MSFD aims to be the environmental pillar of an Integrated Maritime Policy for Europe, which incorporates the Ecosystem Approach and the Precautionary Principle into the management of marine waters.

It is worth also mentioning that there has been a conscious effort to base targets, where possible, on requirements of existing legislation, for example, the Habitats Directive, the Birds Directive, the Water Framework Directive and the Revised Common Fisheries Policy (CFP). However, because of the wider scope of the MSFD, a range of additional targets and indicators have been developed (and, in some cases, remain under development).

Coherence and coordination between the Marine Strategies of each Member State at subregional and regional level are sought through the cooperation of neighbouring Member States (Article 5) and the use of existing regional structures such as the Regional Sea Conventions (Article 6) to coordinate the implementation of the MSFD at regional level. OSPAR, the Convention for the Protection of the Marine Environment of the North-East Atlantic, has such a role for the Northeast Atlantic while similar roles are played by the Barcelona Convention for the Mediterranean, the Helsinki Convention for the Baltic Sea and the Bucharest Convention for the Black Sea.

When assessing the status of selected marine mammal populations against the relevant characteristics of GES, the outcomes could be positive, negative or, as seems to have been the case for most species reported by Member States to date, "unknown" (Fig. 2; EEA, 2014). Nevertheless, the remaining stages of the process need to be completed. Once the main pressures have been identified (in the case of marine mammals these include direct and indirect effects of fishing, maritime traffic, underwater noise, habitat loss, pollution, climate change), if the status of the species, in relation to the criteria provided by the Commission Decision, remains "unknown", the application of the precautionary principle would demand that these pressures be reduced (managed at such a level that there is no risk of adverse impact on population status). If implemented for marine mammals, this would obviously have significant implications for various industries—not least fishing—and hence the potential to generate conflict with various marine and maritime sectors. This leads us to the final key issue, what happens when conservation and other sectoral objectives (framed crudely, environmental versus socioeconomic objectives) are apparently incompatible? Or, to put it another way, can the MSFD succeed where other conservation directives (e.g. the Habitats Directive) have, arguably, failed, i.e. in delivering sustainable use of Europe's seas?

The above quite lengthy introduction to the MSFD is provided to help understand that, although the Directive is well-intentioned, it has a clearly stated general aim (i.e., achievement and maintenance of GES), and has been supported by a fair amount of guidance from the EU, its success is far from assured. The timeline is very tight, the detailed objectives are poorly defined, coordination between Member States is limited, and it is unclear how conflicts between MSFD conservation objectives and the objectives of other sectors (e.g. fishing) will be resolved. In addition, there is an implicit requirement for Member States to substantially increase their current level of environmental monitoring and introduce new conservation measures, without any indication of how these actions will be resourced. Finally, in common with much European environmental legislation, and in marked contrast to US legislation (e.g. the Marine Mammal Protection Act), the MSFD is virtually silent about the mechanisms by which its goals will be achieved. Thus, there is abundant scope for things to go wrong, both the objectives and the timeline are hugely optimistic, and a plethora of implementation issues lies ahead.

Unsurprisingly, the MSFD has already been the subject of a number of reviews and critiques, although De Santo (2010) offers a robust defence of the legislation. Mee et al. (2008) pointed out that the interpretation of ''Good'' in GES is the key to the implementation of the MSFD, but that it evidently relates to human values and worldviews which vary

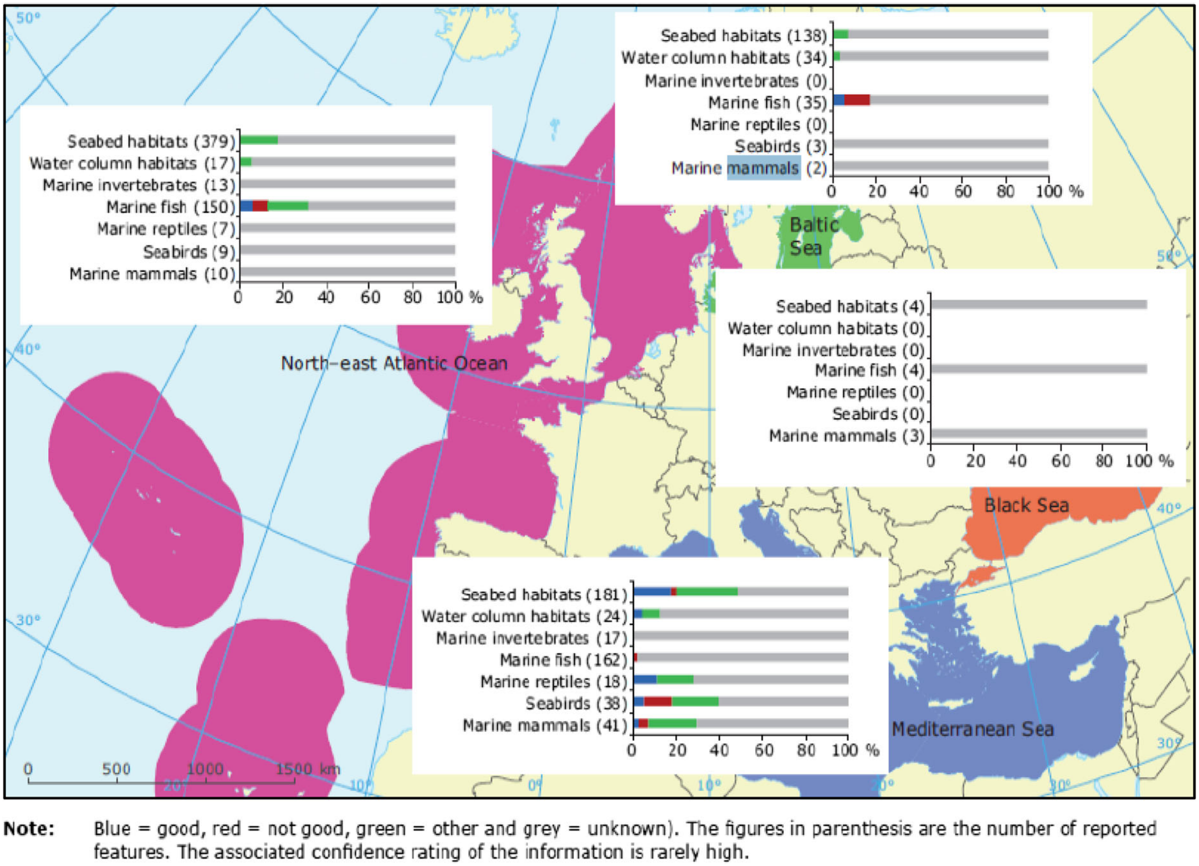

Fig. 2 Status of the natural features reported under the initial evaluation by EU Member States following the requirements set by the Marine Strategy Framework Directive. Reproduced from EEA (2014)

widely across Europe. Van Leeuwen et al. (2012) focused on the "institutional ambiguity" of the MSFD, essentially the lack of clarity as to the division of responsibilities between the EU, Member States and Regional Sea Conventions and as to how the activities of the different actors are to be coordinated. Ounanian et al. (2012) criticised the MSFD for legal vagueness and highlight problems that occur when neighbouring Member States interpret GES differently. Bellas (2014) points out the lack of clarity in the definition of GES, the failure to specify the types of management measures needed and the reliance on existing EU financial instruments, among other things. He also points out that Member States are obliged to "take the necessary measures" to achieve or maintain GES but not to actually achieve or maintain GES!

Suárez de Vivero & Rodríguez Mateos (2012) point to the relevance of the different ways the marine and coastal zone are governed. Thus, Spain has devolved responsibility for management of its coasts almost entirely to the autonomous regions while the central government retains responsibility for offshore maritime areas. In the current paper, we aim to describe how the MSFD process works in practice and to identify issues that have arisen during its implementation, specifically in relation to marine mammals, in particular cetaceans. We specifically refer to the biodiversity descriptor (D1) but some of the associated monitoring would also be relevant to descriptors 4, 8, 10 and 11 (see Box 1). We focus on the experience of a Member State (Spain) where we have been directly involved in the development of the Marine Strategy for cetaceans in each of the 5 sub-regions into which Spanish marine waters have been divided (Fig. 3), and also on the efforts at coordination taking place under the auspices of one of the Regional Seas Conventions (OSPAR) through the work of the ICG-COBAM (of which MBS is a member) and the International

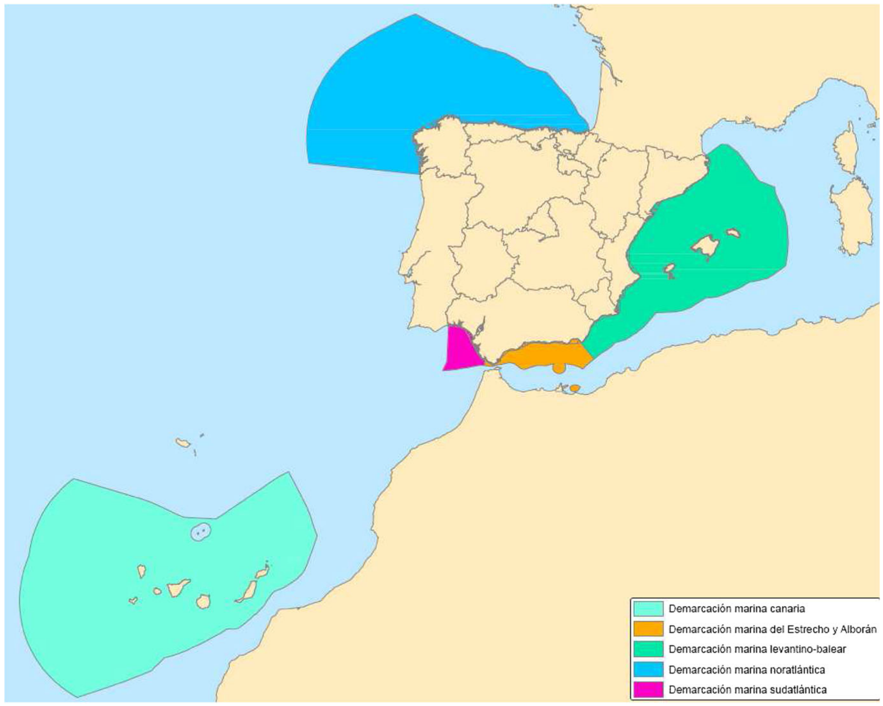

Fig. 3 Maps of the five subregions ("demarcaciones") into which Spanish waters have been divided for the application of the Marine Strategy Framework Directive: the North Atlantic, the South Atlantic, the Canary Islands, the Levantine–Balearic and the Gibraltar Strait and Alborán Sea subregions. Each subregion has its own Marine Strategy. Reproduced from MAGRAMA, (2012a)

Council for the Exploration of the Sea (ICES) Working Group of Marine Mammal Ecology (WGMME) (of which we are both members), which is providing advice to OSPAR. If this may sometimes appear to be an exercise in cataloguing the serial failure of the process to match up to expectations, we also aim to highlight the tremendous efforts being made by many institutions and individuals to implement inevitably imperfect legislation and the significant steps that are being made. We wish to make clear that the opinions expressed are our own (except where otherwise indicated) and are not intended to represent the views of the above-mentioned bodies, or indeed those of any other bodies whose work we discuss.

The process

We describe the process as a series of consecutive steps and for each one mention the science involved, and the challenges (and opportunities) presented by the choices made. The development of a Marine Strategy includes the definition of GES in terms of a series of descriptors including the biodiversity components, and the setting of environmental targets (qualitative or quantitative, with the latter being preferred to facilitate the evaluation of progress) that must be achieved to ensure that GES is reached. A Member State must also develop indicators,[1] that will measure progress towards the achievement of the environmental targets, and reference points (also named "limit values" and sometimes "thresholds"), which should act as triggers for management actions since they will indicate the point along the indicator scale at which GES is lost. Sometimes, targets are also defined specifically for each indicator. In addition, since some targets are defined in relation to a baseline, indicators also need baseline levels (defined as "*the value of state at a specific point against which subsequent values of state are compared*"; see the glossary provided in Supplementary Material Annex 1).

This process is easier to understand with specific examples. Spain defined three environmental targets relevant to marine mammals, the first one being to "*maintain positive or stable trends in populations of key species and top predators (marine mammals, reptiles, seabirds and fish) and in the case of commercially exploited species, keep them within safe*

[1] "*Specific attributes of each GES criterion that can either be qualitatively described or quantitatively assessed*" (Andersen et al., 2013).

biological limits", with population abundance being the indicator proposed to measure progress towards the achievement of this environmental target, and the baseline for the indicator being the current values of population abundance. Figure 4 shows a schematic diagram of how the different components (indicators, baselines—"*the value of state at a specific point against which subsequent values of state are compared*", see glossary, reference points and environmental targets) work in practice, using the example of the indicators defined for marine mammals in Spain.

As a first step, it is necessary to select those species, populations and management units relevant to act as indicators of the environmental status of Member Statés marine waters. Marine mammals in general, and cetaceans in particular, can travel long distances and live in an environment where there are few barriers to their movement. Unsurprisingly, for most marine mammal species, populations [in the ecological sense of demographically (reproductively) isolated units] are wide ranging and, more often than not, extend across waters of more than one Member State. For convenience, "Management Units" (MUs) may be defined, e.g. the part of a population falling within the waters of a member State. However, if an MU comprises only part of a population, its response to threats and to management measures may be hard to predict.

The indicators proposed to measure progress towards achieving GES for this group (see next section) were developed to be applied to the populations of marine mammal species, since ultimately it is the effect on the population, understood as a demographically independent unit, that needs to be evaluated to determine if GES is achieved and to manage the human pressures that are acting on these populations. In cases where the population extends into waters of more than one Member State, it is the combination of the information collected by each country, which will determine the conservation status of the population and the impact of different pressures to which it is subjected within its range. Nevertheless, the MSFD requires Member States to develop their own indicators, monitoring programmes and management measures. Thus, coordination between Member States, in order to achieve coherent assessment, monitoring and management at the level of a marine mammal population, remains as an important challenge.

The earliest stages of the processes were completed by Member States such as Spain before OSPAR published its recommendations; subsequently, OSPAR recommendations have formed the basis of decisions at Member State level. The order in which Spanish and OSPAR decisions are discussed in the following sections reflects this chronology.

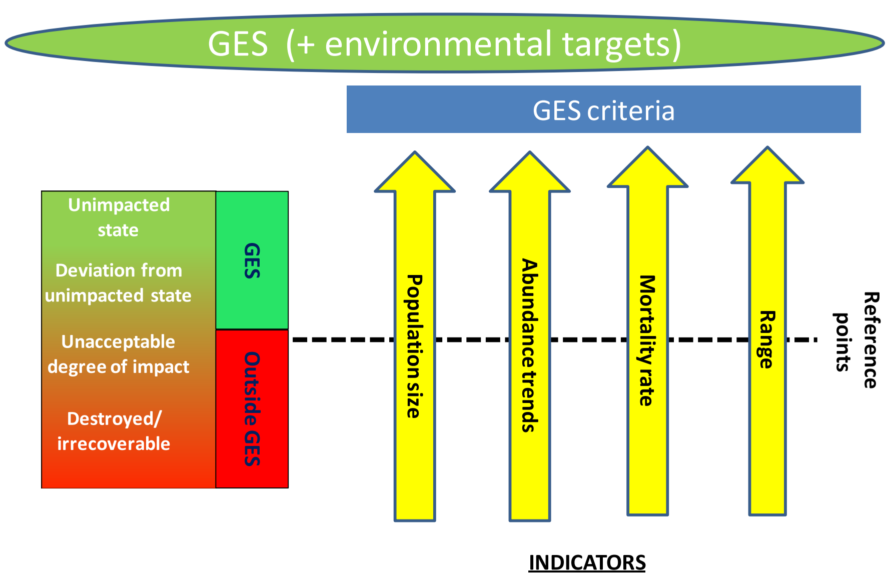

Fig. 4 Schematic representation of the relationship between the different elements design to measure progress help required for the achievement (or maintenance) of good environmental status (GES) as required by the Marine Strategy Framework Directive

Selection of species (and management units)

Spain

The presence of seals on the Spanish coasts is mainly restricted to the appearance of a few individuals of species whose main distribution areas are found in more northern waters, namely the grey seal (*Halichoerus grypus*) and the harbour seal (*Phoca vitulina*), ringed seal (*Phoca hispida*), walrus (*Odobenus rosmarus*), bearded seal (*Erignathus barbatus*), Greenland seal (*Pagophilus groenlandicus*) and hooded seal (*Cystophora cristata*) (e.g. Delibes & Azcárate, 1984; Avellá et al., 1993; van Bree, 2000; López et al., 2002; Alonso et al., 2004; Bellido et al., 2007; Gonzalez-Melcon, 2008; Alonso-Farréet al., 2011; Gutiérrez-Expósito et al., 2012). There are also records of occasional occurrences on Mediterranean monk seals in Spanish waters (e.g. Anonymous, 2008; Font & Mayol, 2009). Taking into account the guidelines for the implementation of the MSFD (Cochrane et al., 2010; EC, 2010), Spain has therefore not included seals in the descriptor on biodiversity.

Thirty-two species of cetaceans have been recorded in Spanish waters (MAGRAMA, 2012b). Not all were considered relevant for inclusion in the initial evaluation, since the occurrence of many is occasional or rare (Table 1), based on the relative frequency of the presence of individuals in the time series of whaling records taken in Spanish waters (e.g. Cabrera, 1925; Aguilar & Sanpera, 1982; Sanpera & Aguilar, 1992; Aguilar, 2006; Aguilar & Borrell, 2007), stranding records for different sections of the Spanish coast (e.g. Pérez & Nores, 1986; Pérez et al., 1990; López et al., 2002; CREMA, 2007; Delphis, 2009; Martín & Tejedor, 2009; Martín et al., 2011; Gutiérrez-Expósito et al., 2012; López et al., 2012) and sightings surveys conducted from shore or from boats and/or aircraft including use of "platforms of opportunity" and dedicated surveys (e.g. Sanpera et al., 1984, 1985; Sanpera & Jover, 1986; Cañadas et al., 2002; López et al., 2004; Cañadas et al., 2005; Gómez de Segura et al., 2006, 2007; De Stephanis et al., 2008; CODA, 2009; Carrillo et al., 2010; Pierce et al., 2010; Spyrakos et al., 2011; López et al., 2012; Hammond et al., 2013).

An expert workshop was convened in May 2014 to ensure that the selected subset of species was representative of the entire cetacean community in Spanish waters and to agree on the baseline abundance levels for the different species to be considered. In almost all cases, the individuals found in Spanish waters are part of populations whose distribution range extends beyond national waters and, in some cases, into oceanic (international) waters. To take this into account, when referring to cetaceans in Spanish waters, the use of the term "management unit" was proposed following the definition provided by the ICES WGMME (ICES, 2014): "*A management unit typically refers to animals of a particular species in a geographical area to which management of human activities is applied. A management unit could be smaller than what is believed to be a population to reflect differences in human activities*". It is important to highlight that it remains necessary to assess the impact of the pressures at the level of the population (or, as defined by WGMME, the "assessment unit"). The use of MUs can be seen as conservative, consistent with the Precautionary Principle, since in general, the local short-term effects of anthropogenic stressors on abundance and distribution are likely to be more severe than the effects on the (usually larger and more widespread) population as a whole.

The criteria agreed for selection of species (actually MUs) were: (a) representation of different ecological niches: coastal-slope waters, oceanic waters, submarine canyons; (b) the existence of absolute abundance estimates, with a degree of precision sufficient to allow detection of population trends (this was not expressed quantitatively; the purpose was more to rule out species for which no or few data exist); (c) priority for other legislation (the bottlenose dolphin and porpoise, for example, are both listed in Annex II of the Habitats Directive) and (d) identification of threats where impacts could be related to the total population abundance (either by monitoring the whole distribution range because the species occurs only in Spanish waters or through collaboration with other countries).

Because, in most cases, the GES of only part of the populations could be assessed in Spanish waters, at least until data collected by other countries could be put together, in the short-term, the work plan outlined in Fig. 5 was proposed. This workplan, based on the application of the Precautionary Principle, involves estimating the potential impact of pressures taking place in national and/or regional waters, on individuals present in those waters. This approach effectively assumes that individuals in these waters would form

Table 1 Cetacean species recorded in Spanish waters and their level of presence based on frequency of sightings and records in the stranding series (for details see main text). Reproduced from MAGRAMA (2012b)

Species	Common name	Presence in Spanish waters
Eubalaena glacialis	North Atlantic right whale	Rare
Balaenoptera physalus	Fin whale	Common
Megaptera novaeangliae	Yubarta	Occasional
Balaenoptera acutorostrata	Minke whale	Common
Balaenoptera borealis	Sei whale	Occasional
Balaenoptera musculus	Blue whale	Occasional
Balaenoptera edeni	Bryde's whale	Common[a]
Physeter macrocephalus	Sperm whale	Common
Kogia breviceps	Pygmy sperm whale	Occasional
Kogia sima	Dwarf sperm whale	Rare
Ziphius cavirostris	Cuvier's beaked whale	Common
Mesoplodon densirostris	Blainville's beaked whale	Occasional, common in the Canaries
Hyperoodon ampullatus	Northern bottlenose whale	Occasional
Mesoplodon bidens	Sowerby's beaked whale	Rare
Mesoplodon europaeus	Gervais' beaked whale	Rare[a], common in the Canaries
Mesoplodon mirus	True's beaked whale	Rare
Lagenodelphis hosei	Fraser's dolphin	Rare[a]
Delphinus delphis	Common dolphin	Common
Lagenorhynchus albirostris	White-beaked dolphin	Rare
Lagenorhynchus acutus	White-sided dolphin	Rare
Stenella coeruleoalba	Striped dolphin	Common
Stenella frontalis	Atlantic spotted dolphin	Common[a]
Stenella attenuata	Pantropical spotted dolphin	Rare[a]
Stenella longirostris	Spinner dolphin	Rare[a]
Steno bradanensis	Rough-toothed dolphin	Occasional, common in the Canaries
Tursiops truncatus	Bottlenose dolphin	Common
Grampus griseus	Rissós dolphin	Common
Globicephala melas	Long-finned pilot whale	Common
Globicephala macrorhynchus	Short-finned pilot whale	Common[a]
Orcinus orca	Orca	Common[b]
Pseudorca crassidens	False orca	Occasional
Phocoena phocoena	Harbour porpoise	Common

[a] In waters of the Canaries subregion

[b] In waters of the South Atlantic and Gibraltar Strait and Alborán subregions (see Fig. 3 for locations of subregions)

part of a local population. The list of the management units selected for Spain is shown in Table 2.

OSPAR

Within OSPAR, the Intersessional Correspondence Group for the Coordination of Biodiversity Assessment and Monitoring (ICG-COBAM) coordinates the biodiversity work related to the MSFD. Expert teams for marine mammals and other functional groups of biodiversity have been set up following nominations by Contracting Parties.

OSPAR proposed guidelines for the selection of the species to be assessed under the MSFD, noting that a coordinated selection would ultimately help to produce comparable assessments and the development of

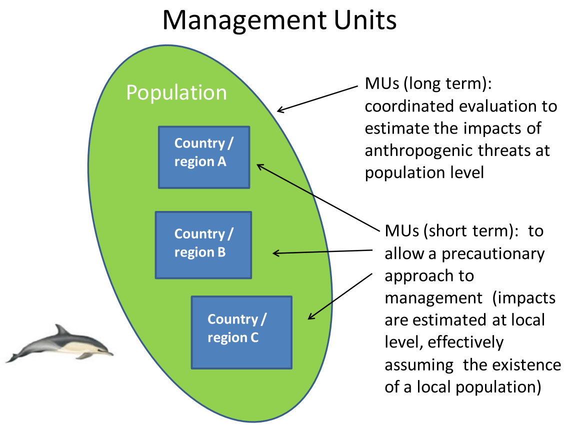

Fig. 5 Schematic diagram of the proposed method to constitute the management units (MUs) for marine mammal populations which good environmental status must be evaluated as part of the Spanish Marine Strategies. Reproduced from MAGRAMA (2014a)

monitoring programmes coordinated between countries. The criteria proposed to guide the selection of species include (1) representativeness in terms of abundance and distribution, (2) sensitivity to specific human pressures, (3) suitability, taking into consideration the indicators and descriptors proposed and (4) feasibility, their monitoring is realistically achievable and/or they are already subject to monitoring and time series of information are available (ICG-COBAM, 2012). The species chosen were harbour seal (*Phoca vitulina*), grey seal (*Halichoerus grypus*), harbour porpoise (*Phocoena phocoena*), bottlenose dolphin (*Tursiops truncatus*), white-beaked dolphin (*Lagenorhynchus albirostris*), minke whale (*Balaenoptera acutorostrata*) and common dolphin (*Delphinus delphis*). This selection implies a focus on shelf waters (with the exception of common dolphins and the offshore oceanic bottlenose dolphins which are known to move in and out of shelf waters) and reflects the importance of these species in the region, the fact that the species selected are also already subject of protection measures under the Habitats Directive, EC fisheries regulations, etc. and the fact that the monitoring and evaluating the distribution and abundance of all marine mammal species would not be feasible, especially in the case of widely dispersed and oceanic species.

The International Council for the Exploration of the Sea (ICES) is an intergovernmental organisation formed by twenty member countries that border the North Atlantic and the Baltic Sea. The main objective of ICES is to "*increase the scientific knowledge of the marine environment and its living resources and to use this knowledge to provide advice to competent authorities*" and ICES achieves this objective through a network of more than 4,000 scientists based in member countries, specifically through their contributions to meetings of Expert Groups. These Expert Groups deliver the science needed to underpin advice on fisheries management, ecosystems and environmental issues, including the management advice requested by member countries, the EU and international organisations and commissions such as OSPAR.

ICES WGMME, acting in response to a request to ICES from OSPAR to advise on appropriate management units for seals and cetaceans, proposed the term "assessment unit" instead of "management unit" for use in MSFD assessments, recognising that these assessments should be undertaken on biologically appropriate units (i.e. biological populations), even if management measures may sometimes be applied on a more local basis (for example in response to a local threat) (ICES, 2014). These assessment units include 5 units for harbour porpoises, 17 for bottlenose dolphins and single units for the remaining species (Fig. 6).

As can be seen, from Table 2 and Fig. 6, despite applying similar criteria, Spain and OSPAR selected different sets of species, although this can be explained in part by the fact that three out of five Spanish coastal regions fall outside the OSPAR area. Nevertheless, because the guidelines in the MSFD are quite vague such discrepancies are to be expected, as different expert groups come up with different interpretations. Ultimately, the MSFD process needs species or groups of species the status of which, in relation to a defined GES, can be followed over time using the selected indicators. One interpretation would be that only those species or groups for which sufficient good quality data exist for *all* of the indicators proposed should be selected, to ensure that evaluation of current status and ongoing monitoring are feasible. Following this interpretation, normally, only common species should be selected, since low densities increase the variability of the estimates and make the detection of trends more difficult. This represents one possible course to take when faced with imperfect data. Another possible course would be to consider that, due to the limitations of the available data to describe the state and the level of pressures, the evaluation of initial status cannot be carried out for

Table 2 Management units for cetacean species proposed in the Spanish Marine Strategy Framework Directive assessments. Reproduced from MAGRAMA (2014a)

Management units proposed				
Species	Subregion	Management unit	Justification	References
Tursiops truncatus	North Atlantic	South Galicia (resident)	Genetics, Isotopes, PhotoID	Fernández et al. (2011a, b)
		N–NW coast (coastal waters)	Genetics, PhotoID	Vázquez et al. (2006), Fernández et al. (2011a, b)
	South Atlantic	Coastal waters	PhotoID (no recaptures), no genetic differentiation	Giménez et al. (2013)
	Gibraltar Strait–Alboran Sea	Gibraltar Strait		
		Alboran Sea	PhotoID	De Stephanis per. comm.
	Levantine–Balearic	Shelf waters (shelf break included)	Genetics	Natoli et al. (2006)
		Balearic islands	Genetics, Isotopes, PhotoID	Brotons per. comm.
	Canary Islands	Canary Islands	PhotoID	Tobeña et al. (2014), Vidal per. comm.
Phocoena phocoena	North Atlantic	Iberian population	Genetics	Fontaine et al. (2007)
Delphinus delphis	North Atlantic	Atlantic	Genetics (lack of differentiation)	Natoli et al. (2006), Amaral et al. (2007), Moura et al. (2013)
	South Atlantic			
	Gibraltar Strait–Alboran Sea	Alboran Sea	Genetics	Natoli et al. (2008)
Balaenoptera physalus	North Atlantic	North Atlantic	Genetics, isotopes	Berube et al. (1998), IWC (2009)
	Levantine–Balearic	Mediterranean		
Orcinus orca	South Atlantic	Gulf of Cadiz + surrounding waters	Genetics, PhotoID	Pérez-Gil et al. (2010), Foote et al. (2011)
Globicephala melas	Gibraltar Strait–Alboran Sea	Gibraltar Strait	Genetics, PhotoID	
		Alboran Sea + Gulf of Vera		
Stenella coeruleoalba	Levantine–Balearic	Western Mediterranean	Genetics	García-Martínez et al. (1999)
Grampus griseus	Levantine–Balearic	Western Mediterranean		Chicote et al. (2013)
Ziphius cavirostris	Gibraltar Strait–Alboran Sea	Alboran Sea + Gulf of Vera	No sightings in the Strait	Di Stephanis per. comm.
	Canary Islands	Eastern islands	PhotoID	Schiavi per. comm.
		Western islands		
Physeter macrocephalus	Levantine–Balearic	Balearic islands	Presence of males + groups	Pirotta et al. (2011)
	Canary Islands	Canary Islands		Aguilar de Soto per. comm.
Globicephala macrorhynchus	Canary Islands	Tenerife–Gomera	PhotoID	Servidio per. comm.
Mesoplodon densirostris	Canary Islands	Eastern islands		
		Western islands		

PhotoID photo-identification

cetaceans (or marine mammals in general) and therefore more data are needed before indicators can be considered fully developed, environmental objectives established and ultimately, cetaceans can be used to evaluate the status of marine waters. This line of argument was used to justify the current non-

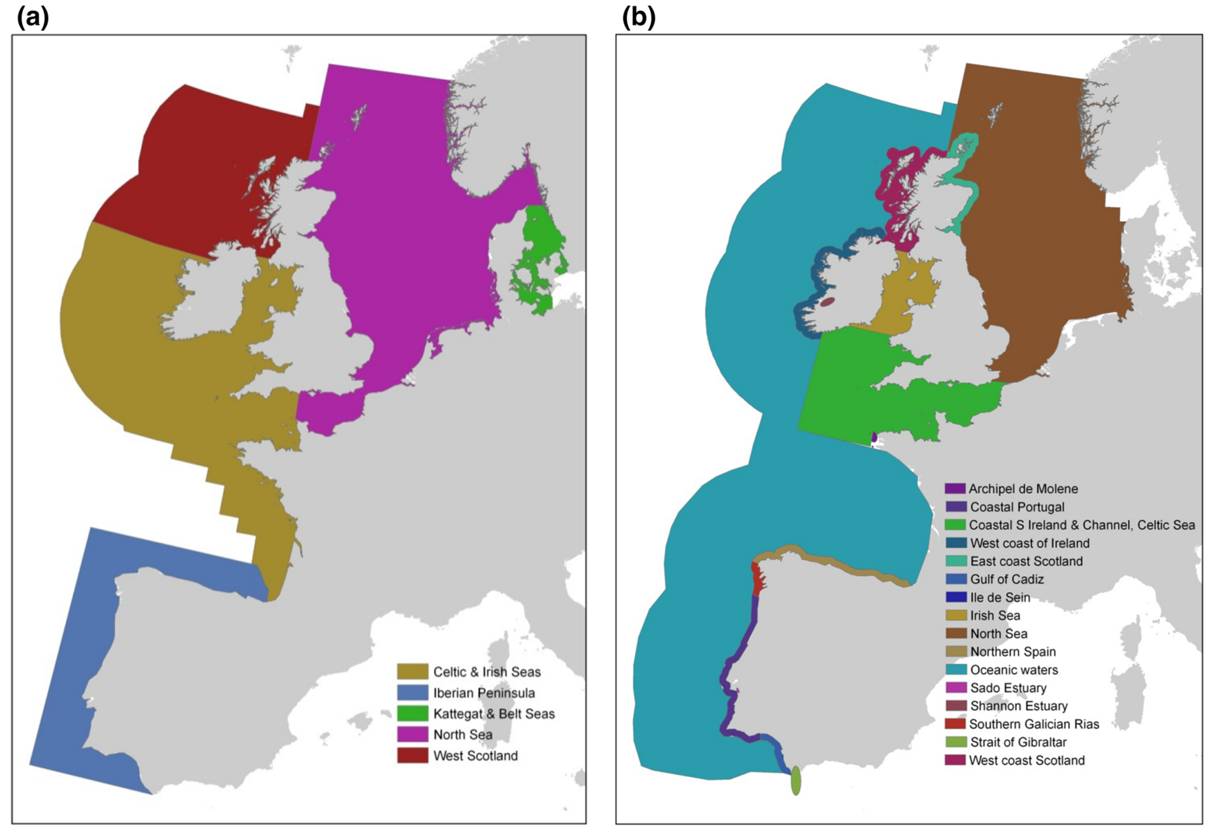

Fig. 6 Assessment units proposed for **a** harbour porpoise (*Phocoena phocoena*) and **b** bottlenose dolphin (*Tursiops truncatus*) by the ICES Working Group of Marine Mammal Ecology (ICES, 2014) for MSFD indicator assessments [Missing from figure b is the proposed Azores bottlenose dolphin assessment unit]. Reproduced from ICES (2014); original maps produced by the UK Joint Nature Conservation Committee (JNCC)

inclusion of marine mammals in the Portuguese Marine Strategy.[2]

The current situation of limited availability of funds for research and conservation (exacerbated by the economic crisis) has led to the monitoring needs created by the MSFD being seen as an opportunity by some to secure new funding together with the concern that funding would only be available for those species and populations selected for the application of the MSFD, with remaining species becoming less of a priority for funding bodies and governments. On the other hand, the high costs associated with monitoring and setting of environmental objectives for cetaceans can also influence the ultimate choice of species since once a species is proposed, Member States are required to determine its status, monitor its trajectory and apply measures to ensure GES is achieved (although, as we will argue below this commitment may be less clear cut than it appears to be). Indeed, the funding required to comply with these MSFD requirements could threaten the availability of funding for basic research and monitoring on other species. In this arena, NGOs and environmental agencies, together with research centres and universities, can also play a part. In a country like Spain, much research on cetaceans has been carried out by NGOs, working on short-term projects supported by regional and national funds, and the inclusion of the species in their region represents an opportunity for continuity of funding, and indeed a means to ensure that their conservations objectives are met. Such considerations may help to explain why, even with a relatively restrictive set of criteria to guide the choice of species, the cetacean experts at the Spanish workshop ended up selecting a wide range of management units, as can be seen in Table 2.

[2] http://www.fc.ul.pt/sites/default/files/fcul/investigacao/DQEM_Continente.pdf.

Indicators

Spain

Spain has followed the monitoring requirements of existing legislation (such as the Habitats Directive), and OSPAR recommendations, and has chosen distribution (range), abundance and demographic characteristics (e.g. mortality rate) as indicators for marine mammals.

Range

The range or distribution of a species/population is the geographical area within which it can be found (Morrison & Hall, 2002) and includes the areas to where individuals may migrate. The Habitats Directive defined range as "*the outer limits of the overall area in which a habitat or species is found at present. It can be considered as an envelope within which areas actually occupied occur as in many cases not all the range will actually be occupied by the species or habitat*".[3] Note that this definition, e.g. with its use of the term "envelope", appears to borrow from the concepts of the fundamental niche (all areas in which a species could occur) and realised niche (those areas in which it currently occurs). However, conservation is logically concerned with the extent of the realised niche, which is typically mainly determined by interactions with other species, resource availability and effects of external stressors. On this basis we should be at least equally concerned about the distribution and occupancy of the "range" as with the extent of the range itself.

In cetaceans, and other highly mobile populations, the presence of a species is much easier to confirm than its absence and spatiotemporal variations in distribution that occur naturally make detecting changes in range difficult—indeed, the accurate determination and quantification of range almost impossible (ICES, 2009, 2014). Further issues include the need to account for seasonal migrations, and the question of how to treat reports of sightings of individuals outside what is considered as the main range of a species; most ecologists would probably accept that occasional excursions beyond the normal range do not constitute a range expansion—although in principle some threshold frequency of occurrence exists above which range extension would be recognised.

Aside from the above-mentioned issue of distinguishing range extension from occasional "vagrant" movements outside the range, detecting a range extension (animals appear where previously there had been no sightings) is relatively simple but the unambiguous determination that a species has disappeared from an area is more difficult. This requires extensive and regular survey effort at and around the limits of the range, something which is rarely likely to be feasible.

The resident populations of certain species in certain areas of the coast, with their small size and limited area of distribution (e.g. resident coastal populations of bottlenose dolphin that have been recorded in Spain and other European countries) represent an exception to this rule and, in these few cases, quantifying range changes should be more feasible (ICES, 2014). The resident population of bottlenose dolphins in the Moray Firth, Scotland is now recognised as having expanded its range to encompass virtually the whole East Coast of Scotland although its core range, protected as an SAC, remains much smaller (e.g. Wilson et al., 2004; Stockin et al., 2006).

Abundance

Abundance is the number of individuals of a population/species in an area. This parameter has traditionally been used as a measure of population health and it is also used under the Habitats Directive to assess the conservation status of populations (as favourable or otherwise based on the trend seen in abundance).[4] Cetaceans are notoriously difficult to study due to their habitat (specifically the high proportion of time spent underwater and, in many species, the low proportion of time they are present in coastal waters), and obtaining abundance estimates at population level generally implies dedicated surveys, complex analysis and high costs. Monitoring conducted during the last two decades has provided the first accurate estimates of cetacean abundance in the European Atlantic

[3] http://www.artdata.slu.se/filer/gybs/notes_guidelines_report_art17_final.pdf.

[4] https://circabc.europa.eu/sd/a/5c427756-166d-4cc8-a654-fca8bfae3968/Art17%20-%20Reporting-Formats%20-%20final.pdf.

coastal and oceanic waters (e.g. dos Santos & Lacerda, 1987; Thompson et al., 2000; Hammond et al., 2002; Cañadas & Hammond, 2008; CODA, 2009; Berrow et al., 2012; Hammond et al., 2013). In the absence of absolute abundance estimates, changes in abundance have sometimes been inferred using abundance indices (e.g. encounter rate) but this is most often at a local scale and population level assessment would require integration of such information across the whole range of a population, ideally based on a common protocol (such as the UK Joint Cetacean Protocol, Paxton & Thomas, 2010). However, currently, such local studies—many of which exist in the different regions of Spain—are rarely planned as part of a larger scale monitoring programme and as such their value, at least for the purposes of the MSFD, is often limited.

Population demographic parameters (e.g. mortality rate)

Knowledge of the demographic parameters of a population (mortality, fertility, health, etc.) can both reveal the current trend in abundance and indicate how the population will change in the future. In principle therefore, this kind of information is useful to detect and assess changes in population abundance caused by anthropogenic pressures. Fishery bycatch has been identified as the main anthropogenic threat to many populations of marine mammals worldwide (e.g. Lee et al., 2006). In Europe, specific legislation (Regulation 812/2004) has been enacted to monitor and mitigate the impact of this threat on cetaceans, at least for certain fleets, although it is likely to be replaced in the near future. Other threats that have the potential to impact on cetaceans populations include collisions with boats, disturbance from the underwater noise generated by boats (also seismic surveys and naval sonar), depletion of their prey caused by fishing, loss or degradation of habitat, pollution, marine debris, disease, the development of renewable energy and climate change (e.g. Tregenza et al., 1997; Tregenza & Collet, 1998; Jepson et al., 2005; Sini et al., 2005; Wells et al., 2005; Hall et al., 2006; Madsen et al., 2006; Northridge, 2006; MacLeod et al., 2007; Wilson et al., 2007; Pierce et al., 2008; Tyack, 2008; Gregory, 2009; Law et al., 2010; Fonseca, 2011; Yap et al., 2012).

The impact of bycatch in Spanish waters is believed to be particularly important in certain areas, notably the NW (Galicia) where by-catch rates could be unsustainable for some populations (e.g. López et al., 2002; Fernández-Contreras et al., 2010; Goetz et al., 2014). However, at a local level, and for different species, populations or segments of the population, other threats may assume greater importance, such as death or injury due to ship strikes, whale watching (fully developed in the Canary Islands and the Strait of Gibraltar and growing in large areas of Spanish Mediterranean waters), etc. To incorporate all these different pressures, mortality rate due to any anthropogenic pressure has been selected as the indicator.

To some extent, the adoption of this indicator is a step into the unknown. Mortality rate estimates can be derived from stranded cetaceans but even in Member States with long-established government-financed strandings monitoring schemes, the use of data collected on mortality has been limited to essentially a sentinel role (e.g. detection of high numbers of by-catch mortalities) rather than providing quantitative estimates of population parameters. One reason for this is obvious, namely the potential biases in strandings data. However, a combination of empirical age data and a model-based approach, incorporating robust assessments of error and bias, has the potential to overcome these limitations (e.g. Winship et al., 2009; Read et al., 2013; Saavedra et al., 2014), especially now that modelling of carcass transport is providing insights into the strandings process (e.g. Peltier et al., 2013). An additional limitation is that strandings are relatively rare in some regions (e.g. the Canary Islands) and as such the mortality indicator may require some revision in these areas.

Other population demographic parameters (e.g. calf production) could also become suitable indicators of population status but their use as indicators is restricted to those populations for which small population size and accessibility could allow the collection of the information needed (e.g. orcas in the Strait of Gibraltar, De Stephanis et al., 2008). This type of information is more difficult to obtain from strandings due to the obvious limitation that healthy mature females tend to be underrepresented, although basing estimates of pregnancy rate on animals that died due to physical trauma can potentially overcome this bias (see Learmonth et al., 2014, for discussion).

OSPAR

The ICG-COBAM proposed six common indicators to be used in the MSFD process for marine mammals. These indicators include range, abundance and the impact of fishery bycatch on populations. The three common indicators specific to cetaceans are as follows:

- Distributional range and pattern of cetacean species regularly present (M-2)
- Abundance at the relevant temporal scale of cetacean species regularly present (M-4)
- Numbers of individuals bycaught in relation to population size (M-6)

As can be seen from this list, there is no mention in the OSPAR common indicators of demographic parameters and only one indicator of pressure (bycatch) has been selected. There is a proposal for an additional common indicator on PCBs in blubber for marine mammals ("Blubber PCB toxicity threshold", ICES, 2014) although it has yet to be agreed.

Again, by trying to cover more pressures in addition to more species, the approach taken by Spain will imply a higher level of monitoring than the one implied by the common indicators developed by OSPAR. However, in both cases, there is a focus on particular anthropogenic threats, with both OSPAR and individual Member States looking to identify those pressures (be it bycatch, or others less easily monitored such as underwater noise, pollution and prey depletion) which need to be regulated to ensure effective conservation of the cetacean populations living in European waters. However, individuals and populations may be adversely affected not only by individual threats, but also by the cumulative and synergistic effect of several threats. The identification of the potential cumulative impacts of multiple human activities has not been integrated within the current monitoring plans, which focus on indicators of individual pressures (as well as indicators of state).

Indicators are ultimately tools to measure progress towards the environmental target, and as such they need a baseline (the point or level with which measurements are compared) and reference points that will mark when the GES is lost. The need for a baseline is most obvious when the evaluation of monitoring data on indicators will focus on the extent, rate or direction of change rather than on absolute values.

Baselines

OSPAR

The ICG-COBAM, in its advice manual and background Document on Biodiversity (ICG-COBAM, 2012) proposed three approaches for the setting of baselines:

Method A (reference state/negligible impacts) Baselines can be set as a state in which the anthropogenic influences on species and habitats are considered to be negligible;

Method B (past state) Baselines can be set as a state in the past, based on a time-series data set for a specific species or habitat, selecting the period in the dataset which is considered to reflect least-impacted conditions;

Method C (current state) The date of introduction of an environmental directive or policy can be used as the baseline state. As this may represent an already deteriorated state of biodiversity, the associated target typically includes an expression of no further deterioration from this state.

As can be seen there have been several methods proposed for the setting of baselines. In the case of marine mammals, the original discussion focused on which levels could be used as representative of unimpacted populations or when, in the long history of exploitation, levels could be considered least impacted. Some experts proposed the use of historical values before marine mammals were considered to have suffered a significant impact due to anthropogenic pressures. For some populations, this could mean the period before the development of the industrial whaling. However, due to the lack of accurate or precise historical abundance estimates (or indeed, any at all) for most cetacean populations, this approach has not been deemed possible and current estimates have been proposed instead, recognising that current abundance values could be lower than the original levels. Another aspect of the rationale has been that it is unlikely that the restoration of populations to these historical levels could be achieved since the carrying capacity of the marine ecosystems could have been reduced.

Spain

For the range indicator, based on the previous experiences with the Habitats Directive, where for many species the favourable reference range has been defined as the full extent of continental shelf waters, Spain has determined that it is not really possible to propose baseline levels (MAGRAMA, 2014a). While disappearance of a species from a significant portion of shelf waters could arguably be detected by a suitable monitoring programme, in many cases it is doubtful that occupancy of the whole of this proposed range has been reliably documented.

In the absence of abundance estimates for unimpacted populations, and with very few available historical estimates before the start of the modern monitoring programmes, Spain has selected current values of population abundance (where available) as the baselines for the abundance indicator (MAGRAMA, 2014a; Table 2).

For the demographic parameters indicator, life history parameters that determine the population dynamics (age structure, birth rate, mortality rate) should ideally be available. However, due to the difficulty of accessing living marine mammals and the fact that dedicated sampling, e.g. of the kind routinely carried out for fish stock assessments, is essentially impossible for a variety of reasons (legal, ethical, logistic, etc), not least that limited data can be obtained from live animals (unless they are individually identified; see below) and lethal sampling would in itself likely represent a significant threat to the populations sampled. Therefore, estimates of these parameters are in most cases derived from stranded and bycaught animals. In those cases where coastal, resident populations have been extensively studied and individual animals identified, it is possible to obtain this information from the monitoring of the individual life history using photographic approaches (e.g. Wells et al., 2005).

Use of stranded animals as a source of samples for the characterisation of marine mammal populations has been extensively discussed in the literature. Issues of concern include the fact that reporting of strandings depends upon the accessibility of the coastline, the density of the human population living near the coastline, the level of public interest and availability of interested organisations/volunteers in the area. Several recent model-based studies have examined factors affecting the likelihood of a carcase arriving on the shore (e.g. Peltier et al., 2012, 2013, 2014). In addition, if the distribution of different age classes and/or of different causes of mortality is spatially heterogeneous in relation to the likelihood of arrival of an animal on the coast, further biases may be expected. At the most simplistic level it may be supposed that strandings will better represent animals which die near the coast. A hidden bias may also be present in terms of which reported animals are subsequently necropsied, related to cost-effectiveness and logistic constraints. Thus, less information can be obtained from badly decomposed carcasses, which may be more likely in areas where monitoring coverage is thin and, consequently, there is a longer interval between stranding and discovery, so they may be excluded in favour of fresher local carcasses. In addition, it may be more difficult and expensive to transport carcasses from remote areas and budgets are likely to be tighter at the end of the financial year. Such biases could also impact on the apparent distribution of causes of death, with causes that are prevalent in remote areas of coastline or towards the end of the financial year being under-represented.

Although life table methodology permits (under certain strong assumptions) translation of the age structure of the sampled dead animals to the age structure of the living population, and hence estimation of mortality rate (e.g. Read et al., 2013), a similar mapping is not feasible for pregnancy rate because achievement of reproductive success is most unlikely to be independent of health status. In the reproductively active component of the living population, the fittest animals are arguably likely to have higher reproductive rates but a lower probability of dying. Basing estimates of pregnancy rate on animals that died due to physical trauma can potentially overcome this bias (see, e.g., Learmonth et al., 2014, for further discussion).

Despite these numerous caveats important quantitative and qualitative information on mortality can be obtained from strandings. Mortality rate will be reflected in age structure and a decline in average age likely reflects increased mortality; even if estimates of absolute mortality rate are biased, changes can be informative. At worst, changes in age structure and/or the frequencies of different causes of death can alert conservation managers to a need for further information on likely causes. Minimally therefore, this monitoring has a sentinel function. Age determination in cetaceans, by counting growth layer groups in teeth, while a relatively straightforward and routine procedure, is time-consuming and length structure could be used as a proxy for age structure. While less likely to be sensitive to changes in age composition of older animals due to the weaker length–age relationship as animals approach asymptotic size (see growth curves in Learmonth et al., 2014), this approach probably could be used to detect changes in mortality of younger animals. Although fewer data are likely to be available to estimate birth rate and estimates from strandings will almost certainly be underestimates, again changes in the estimated rate could be informative.

Environmental objectives

OSPAR

The ICG-COBAM, as it had done for baselines, also proposed three approaches for the setting of targets in its advice manual and background Document on Biodiversity (ICG-COBAM, 2012):

Method 1 Directional or trend-based targets.

i. Direction and rate of change
ii. Direction of change only

Method 2 Targets set at a baseline

Method 3 Target set as a deviation from a baseline

The manual also states in relation to marine mammals that "*taking into account limited data availability for cetaceans, method 1 is advised for target setting, while any of the approaches to set a baseline (methods A, B and C) could be applicable, depending on data and the history of hunting*". In addition, "*target-setting method 1 and baseline-setting method C are advised, building on experience with EcoQOs*".

For each of the common indicators there is a proposed target:

- Common indicator M-2: maintain populations in a healthy state, with no decrease in population distribution with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state.
- Common indicator M-4: maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state.
- Common indicator M-6: the annual by-catch rate of [marine mammal species] is reduced to levels that are expected to allow conservation objectives to be met.

ICES WGMME noted in 2013 (ICES, 2013) that the "targets proposed for all the indicators need to be consistent with the statistical power of existing or realistically feasible monitoring programmes". In 2014 and following this rationale, WGMME recommended that because of the difficulty of proposing baselines, benchmarks and concrete and measurable objectives, to subsume the range indicator (M-2) within the abundance indicator (M-4) and therefore not to conduct independent monitoring for range. The only exception to this recommendation was the small, coastal, resident populations of bottlenose dolphins where changes in range could be more easily quantified and monitored.

Spain

Closely following OSPAR recommendations, in its initial evaluation, Spain established three main environmental objectives for marine species including marine mammals.

- Maintain the distribution range of the species, in such a way that no range decreases are evident that, statistically, cannot be considered as due to natural and climate variability (associated with the indicator for *distribution range*).
- Maintain positive or stable trends in the abundance of populations of key species and top predators (marine mammals, reptiles, seabirds and fish) and in the case of commercially exploited species, keep them within safe biological limits (associated with the indicator for *species population trends*).
- Reduce the main causes of mortality and decline of populations of non-commercial species groups at the top of the food chain (marine mammals, etc.), such as bycatch, ship strikes, ingestion of marine debris, pollution, habitat destruction and overfishing (associated with the indicator for *mortality of species at the top of the food web*)

In addition, another more general environmental objective could also be considered to apply to marine mammals:

- Develop initiatives for the recovery of species and the restoration of habitats when their deterioration compromises the achievement of good environmental status for the biodiversity descriptor (associated with the indicator *conservation status of habitat and species*).

Finally, there is a specific environmental objective dealing with the need for nationally coordinated programmes for marine mammal stranding monitoring, ringed seabirds and bycatch monitoring of marine mammals, seabirds and turtles:

- Establishment of a nationally coordinated monitoring programme of seabird, reptile and mammal bycatch and of marine mammal strandings and ringed seabirds (associated indicator: *existence of the coordination system*).

Reference points

OSPAR

ICES WGMME in 2014 (ICES, 2014) looked at the criteria already available within the international legislation to provide guidance on reference points for the MSFD indicators. Specifically, WGMME looked at the criteria used by the International Union for the Conservation of Nature (IUCN) and the Habitats Directive to determine the conservation status of species. In both cases, these criteria make use of geographical range and levels of population decline to determine if a species/population is catalogued as "critically endangered", "endangered" or "vulnerable" (in the case of IUCN) or if it has a favourable, unfavourable-inadequate or unfavourable-bad conservation status (in the case of the Habitats Directive). As explained in the previous section, due to the difficulties associated with the range indicator, ICES WGMME did not propose reference points (or indeed baselines, etc.) for this indicator.

For the abundance indicator, the Habitats Directive specifies a decline of more than 1% per year (over the reporting period of 6 years) as one criterion to determine the unfavourable conservation status of a species. For the IUCN, the criterion takes into account population generation time and states that a species will be deemed "vulnerable" when there is a "population size reduction of $\geq$30% over any 10-year or three-generation period, whichever is longer (up to a maximum of 100 years in the future)". IUCN sets also quantitative levels of decline for determining "endangered" ($\geq$50%) and "critically endangered" ($\geq$80%) status, respectively. To determine which time period should be used, the generation time of different species can be obtained from the literature (e.g. Taylor et al., 2007) and based on the available data, the criterion to use becomes a reduction of 30% in population size over three generations. ICES WGMME proposed "*population size reduction of $\geq$30% over any 10 year or three generation period, whichever is appropriate to the species concerned*" following the IUCN approach but also taking into consideration that for some assessment units, the three generation period could be dangerously long if population size was small and therefore shorter time scales should be considered. This would be the case for some resident coastal populations of bottlenose dolphins (ICES, 2014). Indeed, in general, the emphasis should probably be on the rate of decline implied by these criteria rather on the timescale over which it is defined to occur, since evaluation of indicator values is presumably intended to be made, with each MSFD reporting cycle (6 years). Even then, in line with the Precautionary Principle, it might be preferable to act based on a lower (e.g. 90%) certainty that a population is declining rather than focus on the rate of decline that can be detected with 95% certainty or the length of time period necessary to detect a decline with 95% certainty. A full power analysis is essential for all proposed monitoring, and some subsequent revision of targets may be required.

For the mortality rate due to bycatch indicator, there are three main methods to calculate the reference point: (a) as a percentage of abundance [e.g. 1.7% of the best population estimates, which has been proposed for harbour porpoise by the International Whaling Commission (IWC) and the Agreement for the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS)] (IWC, 2000), (b) the use of Potential Biological Removal (PBR) and (c) the Revised Management Procedure (RMP) based on the Catch Limit Algorithm developed by the IWC.

These three methods have different data requirements, with method *a* needing an abundance estimate and an estimate of number of bycaught individuals and method *c* using time series on both, population abundance and bycatch estimates to feed a dynamic population model (ICES, 2014). ICES WGMME in 2013 (ICES, 2013) acknowledged that the most suitable method for setting the reference point was method *c*. However, due to the lack of suitable time series for both population trends and estimates of removals due to bycatch, method *a* has been used more frequently. The appropriateness of using a reference point (1.7%) based on a fraction of the best population estimate, which was calculated for the population of harbour porpoise in the North Sea, has been discussed extensively (see ICES, 2013 for further details). ICES WGMME (ICES, 2013) has highlighted the need to calculate reference points for other populations.

Spain

When the initial evaluation was completed, reference points were not established for the range or abundance indicator beyond the general statement that significant declines should be avoided. However, it was also noted that a statistically significant decrease is not equivalent to a biologically significant decline, and it is the latter which should inform the final objective of the monitoring programme (even if in practice constrained by the former).

Spain has followed the ICES WGMME approach and has also concentrated on the abundance indicator, consulting the national legislation to explore appropriate reference points. The Law on Protection of the Natural Heritage and Biodiversity (Ley de Protección del Patrimonio Natural y la Biodiversidad, 42/2007) establishes the basic legal framework for the conservation, sustainable use, improvement and restoration of the natural heritage and biodiversity in Spain. The law creates, amongst other provisions, the List of Wildlife Species under Special Protection, which includes the Spanish Catalogue of Endangered Species. The Royal Decree of 2011 (Real Decreto, 139/2011) develops the List of Wild Species under Special Protection and the Spanish Catalogue of Endangered Species. Of particular relevance for the MSFD are the guiding principles for inclusion of taxa and populations in Endangered Species Catalogues, as approved by the National Commission for Protection of Nature, on the 17th of March 2004.[5] These principles state that for a species to be declared vulnerable, if must meet at least one of the four following criteria, two of which (A and C) refer to the abundance of the population:

A. The population is considered to be declining based on one of the following subcriteria:

   1. It is estimated that the population has declined by at least 20% within the last half century.
   2. Once the effect of the current threat factors has been scientifically evaluated and taking into account the conservation measures in place, it is estimated that decline in the future may be at least 20% over the next 20 years or 5 generations.
   3. An analysis of population viability shows that the probability of extinction in the wild is at least 10% within 20 years or 5 generations, selecting the approach that produces the highest probability of extinction.

C. The population size is below a threshold limit defined by one of the following subcriteria:

   1. The current population is less than 50% of the population size representing favourable conservation status (when this value can be estimated).
   2. The number of mature specimens is less than half the value that the habitats carrying capacity could sustain.

As explained in the previous section, generation time in cetaceans varies between species, being greater for larger species, but values have been proposed of 7.5 years for harbour porpoise (based on unpublished data from Scotland) and about 20 years for a range of other species (Taylor et al., 2007). Following the subcriterion of a population decline $\geq$ of 20% over 20 years or 5 generations in cetaceans, 5 generations would represent periods of 37 years for porpoises and 100 years for other species so 20 years will always be the shorter period. Assuming a constant rate of decline (and hence an exponential decline), a decrease of 20% in 20 years is equivalent to a 1.1%

[5] http://www.circe.biz/files/Criterios%20orientadores%20CNEA.pdf.

reduction in population size per year. Over a period of 5 generations, a decrease of 20% would amount to a reduction of 0.6% per year for porpoise and 0.22% for other species. These rates of decline are thus similar to or less than the rate (1% per year) cited in the Habitats Directive as indicative of unfavourable status, and which is essentially undetectable based on current monitoring of cetaceans.

For the demographic characteristics indicator and specifically for the mortality rate, efforts are directed towards the estimates of the mortality rates that could be considered sustainable based on the development of population models using the biological data derived from strandings (e.g. Read et al., 2013; Saavedra et al., 2014; see also Stolen & Barlow, 2003 and Moore & Andrew, 2008 for examples, of such an approach elsewhere in the world).

In addition, mortality and birth rates can be estimated in small resident coastal populations where individuals are known based on photo-identification, as is the case of for various resident coastal bottlenose populations and other populations around the world (e.g. Gaspar, 2003; Mizroch et al., 2004). In Spain, there is ongoing work in the Gulf of Cadiz to calculate mortality and birth rates in the local populations of bottlenose dolphins and killer whales (e.g. Chico-Portillo et al., 2011).

Monitoring programmes

According to the published timeline for implementation of the MSFD, design of monitoring programmes will be completed in 2014. Monitoring programmes in the MSFD have to be designed to allow the evaluation of population status but also to capture the sources and impacts of anthropogenic threats such as bycatch, noise, etc. To date, the approach proposed for monitoring marine mammals under the MSFD has varied widely between Member States. Thus, the UK has proposed a programme of monitoring not dissimilar to that which already exists. Portugal has so far not defined indicators for marine mammals and thus no monitoring is envisaged. The approach taken by Spain has been to design an idealised programme, hence one which is likely to diminish substantially once funding constraints become clear. At the time of writing, the proposed monitoring programmes for Spain are at the public consultation stage. It is clear that due to the lack of extra funding, most Member States will finally try to make use of available monitoring to fulfil the new requirements under the MSFD.

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ICES WGMME in 2014 (ICES, 2014) discussed the issues that should be taken into account when designing monitoring programs under the MSFD. These include the various factors influencing our ability to detect trends in abundance and mortality/other demographic characteristics of a cetacean population, including the sample size, the statistical distribution of the parameter measured (abundance/mortality, etc.), the magnitude of change that it is necessary to detect and period of time within which the change must be detected. As discussed above, ultimately monitoring must offer the statistical power to detect trends or changes that indicate a departure from (or arrival at) GES. The MSFD allows Member States to set targets; the challenge is to set targets sufficiently stringent that they can genuinely inform conservation, yet not so stringent that monitoring cannot achieve them. To answer such questions, power analysis is needed (see ICES, 2014) but the final decision is necessarily a value judgement, a compromise between conservation objectives and resources available.

Spain

A review of the current monitoring for cetaceans in Spain was undertaken as part of the process of designing monitoring programmes. This review highlighted the fact that a lot of useful information is collected by different agencies in the country, including regional governments, research centres, universities and NGOs. However, the fact that these activities are conducted through local initiatives, which often do not have stable funding, has meant that this information is very fragmented, and although it might be useful within one (or more) monitoring programmes at a larger scale (national for example), it is necessary to coordinate the collection and analysis of this information, ensure that the methodology is correct, that sampling is done in a systematic way and that funding is continued. In addition, gaps in coverage would need to be filled.

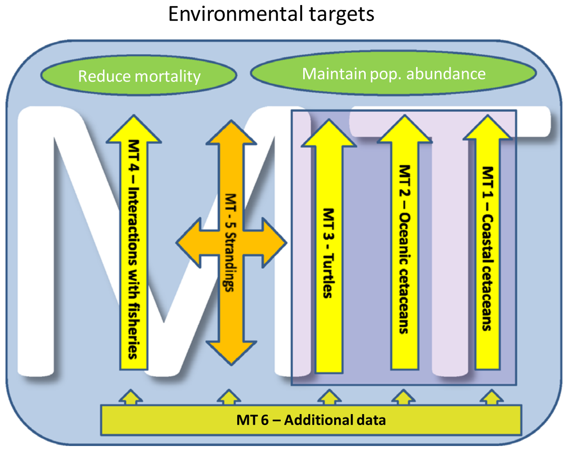

Fig. 7 Schematic diagram of the monitoring subprograms for cetaceans and turtles proposed by Spain as part of its Marine Strategies. Reproduced from MAGRAMA (2014b)

As an example, unlike the UK and France, Spain does not have a nationally coordinated cetacean stranding network. Stranding networks in Spain operate at local and regional scales and are run by different organisations, including universities and NGOs, under agreements with the regional governments. In the case of interactions between fisheries and cetaceans, although a pilot monitoring scheme was run for around 1 year under the auspices of Regulation 812/2004, the current lack of a specific programme to monitor this threat has highlighted the need for the design and creation of a new programme.

Spain has proposed a monitoring programme for marine mammals and turtles that is divided into six subprograms, which aim to obtain the information needed to evaluate periodically, based on the criteria of GES defined for both functional groups, whether GES is reached or maintained (MAGRAMA, 2014b). These subprogrammes follow the recommendations on methodology of ICES WGMME, ICES WGBYC, OSPAR and the Barcelona Convention, and their interrelationships are schematised in Fig. 7. The six subprogrammes are as follows:

– MT-1 Coastal Cetaceans, which focus on the monitoring of abundance of those management units occupying coastal habitats.
– MT-2 Oceanic Cetaceans, also focusing on the monitoring of abundance but of management units for oceanic cetaceans. The division between MT-1 and MT-2 is motivated by the different threats experienced by cetaceans in coastal and oceanic habitats and also due to the different sampling scales needed to cover the heterogeneity present in each of these habitats and, consequently, the different monitoring methods to be used.
– MT-3 Turtles, which focus on monitoring the abundance of turtles.
– MT-4 Interactions with fishing activity, which is designed to quantify the fishery bycatch of marine mammals and turtles.
– MT-5 Attention to strandings, which aims to standardise and coordinate the stranding networks operating in Spain into a single, national programme.
– MT-6 Additional data, which are designed for the coordination, synthesis and integration of many available sources of data (e.g. regular coastal sightings, opportunist sightings, sightings from platforms of opportunity, tagging results, information on habitat use, etc., that can inform the future programme of measures to ensure the conservation of the species/populations.

The remaining stages of implementation of the MSFD

Management measures

The biggest challenges to implementation of the MSFD lie in the near and medium-term future. The proposed monitoring programmes need to be introduced and management measures developed; ideally risk analyses are needed for both. Systems need to be put in place to link monitoring outcomes to management actions and to evaluate the efficacy of both monitoring and management. Decisions will be needed about how to manage the inevitable conflicts between conservation objectives and the objectives of other marine and maritime sectors; appropriate trade-offs will need to be identified based, for example, on some kind of cost-benefit analysis (see Wilson et al., 2009 for a general overview of such considerations).

In relation to interpretation of monitoring results and the design of management measures to achieve (or maintain) GES, there are several obvious challenges ahead.

Firstly, management options for cetacean populations are rather limited. Direct intervention is unlikely to be desirable or feasible. For terrestrial animal populations, interventions may include measures such as provision of nest boxes, predator control, and vaccination against disease—however, even apparently benign interventions can have unintended consequences, due to the obvious fact that species are embedded in complex communities. Using a model-based approach, Chauvenet et al. (2011) showed that vaccinating lions in Tanzania against canine distemper virus could lead to extinction of the local cheetah population. In practice therefore, management for cetacean conservation under the MSFD will be focused on managing the human activities that threaten (or potentially threaten) cetacean populations, such as fishing, shipping, marine energy development, pollution, and so on. In short, in general the only feasible and effective management measures are those which limit or remove anthropogenic threats. Possible exceptions relate to measures designed to improve habitat quality, e.g. by allowing prey abundance to increase to optimum levels.

In practice, at least in Europe, existing conservation measures for cetaceans are rather weak. Indeed many conservation plans consist of little more than monitoring programmes. Essentially, this is the "easy" option. Well-designed monitoring programmes can be put in place without the need to confront other interest groups whose activities would be affected by any actual conservation measures. Under the Habitats Directive, Special Areas of Conservation have been designated for a number of resident coastal bottlenose dolphin populations. However, these and other types of marine-protected areas may afford little or no protection in practice, especially for highly mobile species which freely move across and beyond reserve boundaries.

The main measure introduced to reduce fishery bycatch of cetaceans in Europe has been the use of "pingers" (acoustic deterrents) attached to nets in some gillnet fisheries (as introduced under Regulation 812/2004) with the aim of reducing harbour porpoise mortality. Even in these cases, the efficacy of the technique is not well-monitored, and indeed it is unclear whether many of the pingers in use are actually operational. In most fisheries that are known or thought to cause significant by-catch mortality in cetaceans, no relevant management measures are in place. Possibly, the best approach will be to involve fishermen in developing solutions (see Goetz et al., 2014), through co-management; what is almost certain is that measures introduced without consultation and without buy-in from the industry will be doomed to fail.

Much has been written about the importance of co-management in fisheries but it is also true that when the alternative appears to be financial ruin, fishermen almost invariably (and logically) take a stance against the imposition of conservation measures, even if the conservation measures are, in principle, in their own long-term interest. Such risk-prone behaviour is predictable under economic and behavioural theory (e.g. Kahneman & Tversky, 1979; Denrell, 2007). The current situation of many fish stocks is a case in point. Drastic cuts in the Total Annual Catch limits have been ordered when stocks are shown to be depleted, but fishermen's representatives bemoan the unwillingness of governments to consider alternative measures. If fish stocks can be rebuilt (and reducing fishing mortality is normally the only plausible mechanism), and fishing is again profitable, imposition of additional conservation measures could be less problematic.

As has been seen with the Habitats Directive, the requirement to mitigate or eliminate fishery bycatch of marine mammals is ultimately limited by the legal priority afforded to the CFP (Fock, 2011; Leijen, 2011; Proelss et al., 2011) and indeed the reality that fisheries interests are usually treated more seriously than conservation needs. The strength of the MSFD in relation to other marine legislation remains to be tested.

Management measures will likely need to be worked out not just with the fishing industry but with the shipping industry (especially in relation to fast ferries which cause significant mortality, for example, of sperm whales in the Canary islands) (e.g. Arbelo et al., 2013), and the renewable energy and oil and gas sectors, among others. The introduction of the forthcoming EU Marine Spatial Planning Directive may help provide a mechanism and rules of engagement for such negotiations.

Generic issues and closing remarks

The EU probably needs to do more to secure the monitoring programmes necessary to meet MSFD requirements, especially for mobile species such as cetaceans, the conservation of which depends on international cooperation and coordination. Perhaps the single most important component of European cetacean monitoring programmes, the proposed SCANS III survey of cetacean abundance in European Atlantic shelf waters (and presumably its successors, proposed to take place on a 6–10-year cycle), will happen or not based on the outcome of a competitive funding application.

What has been apparent during the development of proposed monitoring for marine mammals in Spain is the high level of engagement from a range of NGOs, universities, research institutions and regional government bodies. While the need for coordination, and indeed more funding, is also apparent, a substantial human resource is available to help implementing the MSFD.

It is important to recognise that conservation is in many ways no different to any other human endeavour and conservationists would do well to heed lessons from fields such as psychology and management science, for example, on the application of game theory to human behaviour and on the unintended and sometimes perverse outcomes of management by targets. Frank & Sarkar (2010) argue that the use of game theory, to help understand how different actors will respond to different measures, is essential to achieve conservation goals, although interesting they finally point to the importance of reciprocal relationships of trust.

Any legislation based on targets invites a focus on the targets rather than what they are supposed to measure. As budget and other practical constraints become evident, there will be inevitable pressures to modify objectives, criteria, etc, so that GES increasingly looks like the status quo.

One might reasonably ask whether GES will be more easily attained by strict management of human impacts or by redefinition of the concept to more closely match the current state of the seas. This is not an unrealistic proposition. The UK has "National Parks" that meet few of the internationally recognised characteristics of such areas. Another feature of terrestrial conservation in the UK has been the so-called "Green Belt", designed to prevent cities sprawling into the surrounding countryside. In 2003, Aberdeen City Council proposed to retain the "Green Belt" around the city, but re-defined the concept so that it allowed building houses. The plan was finally abandoned but the latest Structural Plan argues that the Green Belt does not necessarily have to be a 'belt': "*the intention is to… shape Aberdeen into a series of green fingers and wedges with new communities within*". These observations are not intended to imply that anything especially wrong or untoward is happening, rather that irreconcilable objectives tend to produce imaginative solutions that are not what those writing conservation legislation had in mind.

The current move towards integrated ecosystem assessment and management in fisheries (as endorsed and embraced by ICES) has the potential to increase the focus on the conservation measures needed to achieve sustainable resource use (although, as in the case of the MSFD, financial constraints may provide a (reality) check on ambitions in this direction). The MSFD may be seen as advancing the European Community's stated aims of implementing a precautionary and ecosystem-based approach to marine environmental management. A number of factors have been identified as hampering progress in this direction. In addition to highlighting the power of lobby groups, De Santo (2010) discusses the issue of uncertainty, arguing that its value in the decision-making process is misunderstood by regulators. Pointing out that the requirement (under the CFP) for 'evidence of a serious threat' before action could be taken is contrary to the Precautionary Principle; she concludes that "*a lack of certainty should be a call for gathering more information, not for hesitation or inaction*".

In the not too distant future, the introduction of Integrated Maritime Policy (IMP, incorporating integrated marine management and marine spatial planning, MSP, among other related concepts) in European waters (see Qiu & Jones, 2013) may help to clarify the priority afforded to conservation compared to other marine and maritime sectors. As a first step, In July 2014, the European Parliament and the Council adopted legislation to create a common framework for maritime spatial planning in Europe.[6]

Many commentators have argued that IMP/MSP will reduce the influence of the fishing industry [the introduction of the MSFD already implies that the "*sole competence of the CFP to manage fish resource*

[6] See http://ec.europa.eu/maritimeaffairs/policy/maritime_spatial_planning/index_en.htm.

conservation issues is terminated" (van Hoof & van Tatenhove, 2009)]. However, Suárez de Vivero & Rodríguez Mateos (2012) argue that the immense efforts (and resources) required to implement the MSFD in Spain and similar countries will preclude implementation of any other aspects of IMP for at least 5 years.

When IMP is finally implemented, doubts exist as to whether the environmental focus of the MSFD will remain centre stage or whether IMP will re-establish the primacy of economic objectives (see Ounanian et al., 2012 and references therein). These authors further observe that the various marine and maritime sectors (fisheries, offshore renewable energy, offshore oil and gas, navigation, coastal tourism and non-industry stakeholders such as environmental Non-Governmental Organizations) are not all on an equal footing in relation to the MSFD, differing widely as they do in their "*institutional capabilities, economic strength, and political clout*"; they conclude that the MSFD alone will not solve all conflicts between users (and conservers) of the oceans.

Finally, the challenge of implementing the MSFD is one of governance, what van Leeuwen et al. (2012) described as the "*multi-level governance dynamics of EU policy*". Van Hoof et al. (2012) argued that the processes of regionalisation and integration of policy (implicit in ecosystem-based approaches to marine management, such that set out in the MSFD) require further development of the marine governance system, specifically "*positioning the regional level into the multi-level governance system*".

It may be necessary to accept that the search for governance solutions which can achieve sustainable resource use is a "*wicked problem*" (Jentoft & Chuenpagdee, 2009), one which resists all attempts at solution. However, while we do not believe that GES will be achieved by 2020, by fostering the development of a comprehensive set of environmental objectives and, potentially, putting in place enhanced marine monitoring programmes, the MSFD will at worst strengthen the scientific evidence base and help to improve the governance of marine conservation.

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