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13–20 April

Copenhagen, Denmark

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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Executive summary

The Terms of Reference for WGECO in 2011 were more diverse, and also more focused on responses to other groups within ICES than has been the case in some previous years. There was also a considerable overlap in scope between the ToR. As in previous years, there was considerable focus on the science needed to support the objectives of the Marine Strategy Framework Directive (MSFD), and in particular on biodiversity, but also in terms of other descriptors. The other theme that ran through the work was the use of spatially explicit approaches, and in particular the difficulties in establishing Marine Protected Areas in a changing climate.

- The first term of reference (ToR a) continued the work conducted by WGECO over a number of years in developing the Large Fish Indicator (LFI) and its use outside the North Sea. LFI analyses for the Celtic Sea, and initial approaches to developing LFI for other fishery areas are presented together with theoretical studies of LFI recovery.
- The second ToR was planned to address Integrated Ecosystem Assessment (IEA; also discussed in 2010), but it was decided to defer this ToR for one year pending reports from IEA Expert groups (WGNARS, WGIAB, WGINOSE and WGEAWESS).
- In the third ToR, WGECO has examined the report of the Study Group on MPA networks, and made several recommendations. In particular, that network designs should pay particular attention to MPAs around hydrographical range boundaries, and that we need a greater understanding of the 'behaviour' of species of particular conservation concern in the face of climate change.
- The fourth ToR was generic, developed by SCICOM and ACOM to determine how EG could assist in the MSFD and its Good Environmental Status (GES) process. This has been a major field for WGECO for some years and this work is summarized in this report, as WGECO felt that it was important that the work described in past WGECO reports should not be overlooked. We also provided guidance on the definitions of thresholds, targets and the terminology used in the various policy drivers. WGECO also provided information on the approaches to the MSFD and specifications on GES being developed in a number of different countries, as well as a response to the work of WKCATDAT looking at the role of surveys in collecting MSFD data.
- A specific and detailed examination of the issues surrounding GES for biodiversity was addressed in the context of ToR e that considered the ICES Strategic Initiative on Biodiversity. A detailed examination was made of the trade-off between management of biodiversity and sustainable use, and whether biodiversity goals should be set as limit or threshold reference levels. Another issue was the likely need to adjust biodiversity reference levels in the context of changing conditions. In all cases the complexity of issues precludes the development of fully formalized approaches and raises the need for using expert knowledge or "expert input"; a structured procedure for making use of expert input was also examined.
- The final ToR (f) looked at issues linked to marine spatial planning, human pressures and biodiversity, and was also in response to a generic ToR from

SCICOM and ACOM. In particular, WGECO examined the report of WKCMSP and developed some of the recommendations and issues. We provide a view on the link between changes in both human pressures and biodiversity in relation to marine spatial planning. The focus lies on the large development plans for offshore renewable energy and on relevant information for the development of pressure indicators in relation to biodiversity indicators such as habitat biodiversity. We also identified some of the general gaps in spatial data and analysis to support area based management regimes such as marine spatial planning.

A more detailed summary of the work carried out under each ToR is presented below.

In ToR a, WGECO addressed the need to establish a consistent process for applying the Large Fish Indicator (LFI) to marine regions outside the North Sea. This required both a technical protocol and additional research into the properties of the LFI as a management tool. Key issues identified were the need to define an appropriate region-specific fish species complex and a corresponding "large fish" length threshold. In particular, the chosen complex must be robustly sampled by the survey gear, and should typically include species that represent key functional roles in the demersal fish community.

LFI for the North Sea (WGECO ICES, 2010), Celtic Sea and Baltic Sea all have lags of approximately 10–15 years in relation to changes in fishing pressure. This may suggest that the lag is a generic phenomenon that should be accommodated when using the indicator for management advice. Recent developments in process-based modelling may well prove useful in understanding the underlying mechanisms for this lag, and offering meaningful predictions of future states. Following ICES (2010), some discussion of new models and their use is provided. While outputs from these models suggest recovery periods of several decades, recovery trajectories asymptote in an exponential form, implying that initial recovery rates may be higher, so that some improvements occur over short time-scales, although full recovery to a new "equilibrium" state may take considerably longer.

Finally, WGECO commented on the use of the LFI as a "foodweb" indicator within the MSFD. It is concluded that the LFI, as a size-based metric, may function well as an indicator of marine foodweb structure. However, WGECO suggests that further research is required so that this can be clarified before the first review of the targets and indicators of the MSFD in 2018.

In ToR c, WGECO examined the work of SGMPAN and concluded that there were two areas where further developments were appropriate and relevant to both WGECO and SGMPAN.

The need for MPA networks has been reasonably well established. However, fishing and other anthropogenic activities may precondition a system in such a way that the rate of movement and the establishment of new colonizers may be altered. MPAs may relieve pressures on new colonizers. A network that is resilient to the effects of climate change may be achieved through protecting representative habitats along the expected change gradient. Physical oceanographic climate-induced changes in substrata (e.g. mediated through storm events in coastal areas) or circulation (e.g. movement of the Gulf Stream) would influence direction of colonization, while other factors such as temperature and salinity changes would influence the timing of colonization events. To a large extent a sufficiently extensive and global network will alleviate many of these concerns.

It is now widely recognized that marine invertebrates often have short realized dispersal distances, for example compared to pelagic fish larvae. Therefore, precautionary design criteria should assume connectivity through larval dispersal over appropriately short distances. While the local networks provide for redundancy and protection against unpredictable events (oil spills, disease outbreaks) there should also be strong connections. While we might envisage a gradual movement of taxa in the face of changing environmental conditions, hydrographic features and current biogeographic boundaries might act as barriers to this movement until they undergo sudden catastrophic change, and should be a particular focus for MPA networks. WGECO also felt that there was a need for a greater understanding of the 'behaviour' of species of particular conservation concern in the face of climate change.

In ToR d, WGECO addressed how its work could support the development of the MSFD descriptors and of the definition of Good Environmental Status (GES).

Several previous reports of WGECO have provided material and discussion that support the indicator selection and target setting processes that are required for the MSFD. A comprehensive catalogue of the relevant work of WGECO was included as an annex, and this was also set in the context of other relevant directives (e.g. Water Framework Directive, and Habitats Directive). Many of the points which referred to the selection of thresholds and the relationship between pressures and target achievement were reviewed and further developed. Biodiversity issues were addressed under ToR e.

In the second part of the ToR WGECO produced a summary of current European Member State (MS) approaches to GES assessment, and a review to show existing best practice in the MS, and in other fora e.g. OSPAR. In addition, a comprehensive review of relevant current research was carried out, and suggestions made for the best tactical use of existing knowledge, as well as for more strategic development. The issues surrounding the monitoring needs for the MSFD were highlighted in the ToR. The principal vehicle for such monitoring is likely to remain the research vessel survey programme carried out by member states. The potential for the surveys to provide wider ecosystem data was documented by ICES WKCATDAT from the perspective of the data provider. WGECO provided the perspective of the data user, and indicated priorities and particular issues in the collection of monitoring data.

The final part of the report deals with the lessons learned thus far, and offers guidance on best practice.

In ToR e, WGECO focused on the important issues involved in the determination of GES in the context of biodiversity. This comprised two elements. In the first we examined the issue of the trade-off between biodiversity conservation and sustainable use. In the second we considered the process of setting biodiversity targets and reference levels in the context of our limited knowledge and of climate change.

Although high-level objectives for biodiversity conservation and fishery management are consistent, any level of harvest will likely affect the size structure, species composition, and biomass of the community, impacts that will be reflected in biodiversity indicators. Thus, any management strategy will involve some level of trade-off between biodiversity and sustainable use. WGECO considered several examples of this trade-off for model and empirical results. The trade-off frontier is typically curvilinear, such that a small reduction in sustainable yield near the maximum may translate into a larger increase in biodiversity indicators. Simulation studies have been used to identify harvest policies that preserve most of the biodiversity while maintaining most of the value of the fishery. In the case of depleted communities, the short–term conservation and sustainable-use objectives are strongly aligned when both prescribe rebuilding; it is only after rebuilding occurs that incompatibilities may become more explicit.

The trade-off between biodiversity conservation and sustainable use can be mitigated, not eliminated, with the choice of management measures. Certain gear types are known to have greater ecosystem impacts than others. Gear technology is used to improve size and species selectivity, and to reduce the impact on non-target species and habitats. However, selective fishing may not be the best strategy to protect biodiversity if it alters size composition and community structure. Spatial management involves an explicit trade-off between fishing opportunities and the protection of habitats and other components of biodiversity. This trade-off is particularly strong for deep-sea fisheries and other habitats with slow growing and/or fragile fauna. Quantifying this trade-off and the benefits of spatial management requires knowledge of the degree of overlap between fisheries and vulnerable habitats and species, and valuation of the costs and benefits.

WGECO notes that reference levels for sustainable use are generally set as limits, whereas biodiversity goals are expressed as target levels or directions. WGECO has previously detailed the conceptual link of the notion of sustainable use with a system's ability to recover rapidly and securely from pressures that are applied. In principle, this analysis can be applied to assess when components of biodiversity are subject to irreversible harm, but there are few examples in which recovery capacity has been measured in relation to biodiversity, and rarely in the context of multiple pressures. WGECO identified several situations in which expert input is needed to identify reference levels and to determine the status of biodiversity indicators or ecosystems. WGECO would recommend the use of a structured procedure for obtaining and documenting expert input and outline such a procedure relevant in the context of the MSFD.

Finally, it is recognized that it may be seen as necessary to adjust reference levels for biodiversity indicators in response to changing conditions, especially climate variation. Procedures for adaptively changing reference levels are well understood for single-species management, but the criteria for adaptive management are unlikely to be met in a biodiversity context. Given the difficulty of identifying reference levels for biodiversity in the first place, WGECO does not recommend a procedure for adaptively changing reference points at this time.

In ToR f, WGECO examines the WKCMSP report and looks at the issues surrounding the development of marine integrated management using marine spatial planning (MSP). These include the implementation of risk-based decision-making and the quantification of uncertainty in the planning process. Risk-based decision-making in spatial planning is related to the importance of providing the science base for activity-pressure-state relationships. Previous work by WGECO focused on the use of the activity-pressure-state relationship in integrated assessments, and this is discussed in relation to the development process of marine spatial plans. Based on previous work of WGECO in 2010 some generic pressures related to offshore renewable energy are listed together with a review of development plans for offshore renewable in Denmark, Germany and the UK. Changes in human pressures at different scales and related changes in habitat biodiversity are described with a hypothetical example to provide information on how to derive related pressure indicators.

WGECO is probably not the right group to provide spatially resolved data; however, we have identified some of the gaps in the availability of spatially resolved data and its application to MSP. Recommendations are made on how ICES can improve the data provision to support MSP and build on existing data infrastructure.

A common theme of the topics discussed was that they generally dealt with the bigger picture and focused on methods, transboundary/regional questions and the need for frameworks to deal with logistics and on MSP at a regional/transnational scale. The ability of ICES to support area based management such as MSP is limited to the provision of specific scientific advice and the provision of spatially resolved data. As outlined in the WKCMSP report, the planning process involved in MSP is based on interaction between policy, managers and stakeholders, with the scientific community having a data provision function; however ICES could have a role in:

- providing experience and networks to facilitate regional assessments, focusing on dealing with MSP in transboundary/regional seas contexts;
- providing a science base to define activity-pressure-state relationships to support risk based decision-making in planning processes;
- evaluation of ecosystem goods and services, which can then be assessed in relation to trade-offs within MSP processes;
- evaluating concepts such as carrying capacity in relation to the acceptable degree of change in the state of habitat biodiversity due to pressures from large renewable energy developments. This includes the assessment of local impacts and the extrapolation to larger scales in the absence of empirical data.

A critical weakness is the lack of methods for assessing cumulative or combined impacts that take account of additive, synergistic and antagonistic effects.

1 Opening of the meeting

The meeting was opened at 10.00 am on 13 April and adjourned on 20 April 2011. The meeting was chaired by David Reid, Ireland, and attended by 27 participants from 13 different countries. Two of the participants contributed by correspondence. A full participants list is found at Annex 1.

2 Adoption of the agenda

The agenda was considered and it was agreed to delay agenda ToR b until the 2012 meeting. The draft agenda is found at Annex 2.

3 ToR A: Application of the Large Fish Indicator

a) Provide guidance on the use of the proportion of large fish indicator in areas outside the North Sea.

The Large Fish Indicator (LFI) was developed over some years to support the OSPAR EcoQO related to 'fish communities', a process described in the 2010 WGECO report and comprehensively in Greenstreet *et al*. (2011). During the selection of indicators to describe good environmental status under the MSFD, the LFI has subsequently been identified as one indicator of 'foodweb structure'. Having been developed almost exclusively in the North Sea (OSPAR region II; MSFD Greater North Sea subregion), the LFI must now be applied to a range of other marine regions. Because these areas are likely to show differences in both underlying ecology and data availability, WGECO recognized that some local tuning of the LFI may be demanded in each new case. A flexible protocol for this tuning process, based on the work of Shephard *et al*. (in review) and focusing on the Celtic Sea, is described below. This protocol is then applied to three additional marine regions: the Grand Banks, eastern Baltic and Biscay. However, none of these represents an entire OSPAR region or MSFD subregion. In fact, the North Sea is the only area at either of these administrative scales that is covered by a single bottom-trawl survey, and even here the first quarter International Bottom-trawl Survey (Q1 IBTS) on which the indicator is based does not cover the whole of OSPAR region II, which includes the Channel. Here we consider the situation for OSPAR Region III (waters west of the British Isle to the shelf edge) where we start the process of developing individual LFIs using several surveys, each of which covers a small part of the region, but which combined cover most of the OSPAR region.

As the ecosystem approach to fisheries management becomes operational, indicators like the LFI will be used as the basis of management advice, e.g. through the Common Fisheries Policy. An issue that has emerged during the development of the LFI and its current application to new regions is the existence of a time-lag of 10–15 years in the response of the LFI to changes in fishing mortality. This has obvious implications for the delivery of short-term advice and indicates the need for robust process-oriented models to support this process. Available models were discussed in ICES (2010), and more attention is given to them here. In particular, an in-depth modelling study on the recovery of the LFI subsequent to reductions in F is summarized. This analysis indicates recovery periods of several decades, and suggests that full recovery may never occur in areas that have been very intensively fished for 25–50 years.

As mentioned above, the LFI has been identified within the MSFD as an indicator of good environmental status in 'foodweb structure'. However, it is not clear how well the LFI will fulfil this role. WGECO here discusses this issue and recommends further consideration.

3.1 A protocol for developing regional LFI indicator series

This section summarizes a study by Shephard *et al*. (in review), which can be referred to for further detail.

The Large Fish Indicator (LFI) has been developed as a univariate indicator of the effects of fishing on fish community 'state'. The LFI describes the proportion (by weight) of the fish community that is larger than a specified length threshold (40 cm for the North Sea; Greenstreet *et al*., 2011). The LFI provides a metric that can be related to a defined Ecological Quality Objective (EcoQO) based on a reference period when the fish community in question was considered to be sustainably exploited (1983 in the North Sea, giving an EcoQO of LFI≥0.3). The LFI has been adopted as the 'fish community' EcoQO for OSPAR regions and as a 'foodweb indicator' in the EU Marine Strategy Framework Directive (MSFD). This will require application of the indicator to widely varying marine ecosystems, fish communities and survey protocols. Reflecting the fact that fish communities vary between regions, this requires the adaptation of the method developed in the North Sea to ensure the full utility of the LFI, and the definition of appropriate LFI targets in each new application.

The rationale underpinning the North Sea LFI has been interpreted and applied to the development of an LFI specific to the Celtic Sea and to the definition of an appropriate management target for the region (Shephard *et al.*, in review). The principal issues identified in this adaptation focused on:

- Which species should be included in the species suite?
- At what length threshold should fish in the community be considered "large"?

In the North Sea, Norway pout were included in the species suite, being deemed to be adequately sampled by the survey trawl gear. The North Sea LFI was equally influenced by both changes in the biomass of small fish and changes in the biomass of large fish. Changes in small fish biomass were primarily the consequence of the indirect effect of fishing; the removal of large fish predators, reducing predation mortality on small fish, and allowing the expansion of their populations. The Celtic Sea is a recognized nursery area for blue whiting (*Micromesistius poutassou*) and in this region, blue whiting may well fulfil the same ecological function as Norway pout in the North Sea. Consequently blue whiting, although excluded in the North Sea species suite, were included for the Celtic Sea analysis (Figure 3.1.1). In the Celtic Sea, boarfish (*Capros aper*) form large pelagic shoals, such that variation in their abundance would be inadequately represented in Celtic Sea survey samples. As such, inclusion of boarfish in the Celtic Sea LFI analysis would simply have added to "noise" in the metric trend, and hence this species was excluded. In the North Sea, boarfish are so infrequently encountered that they were included simply by default.

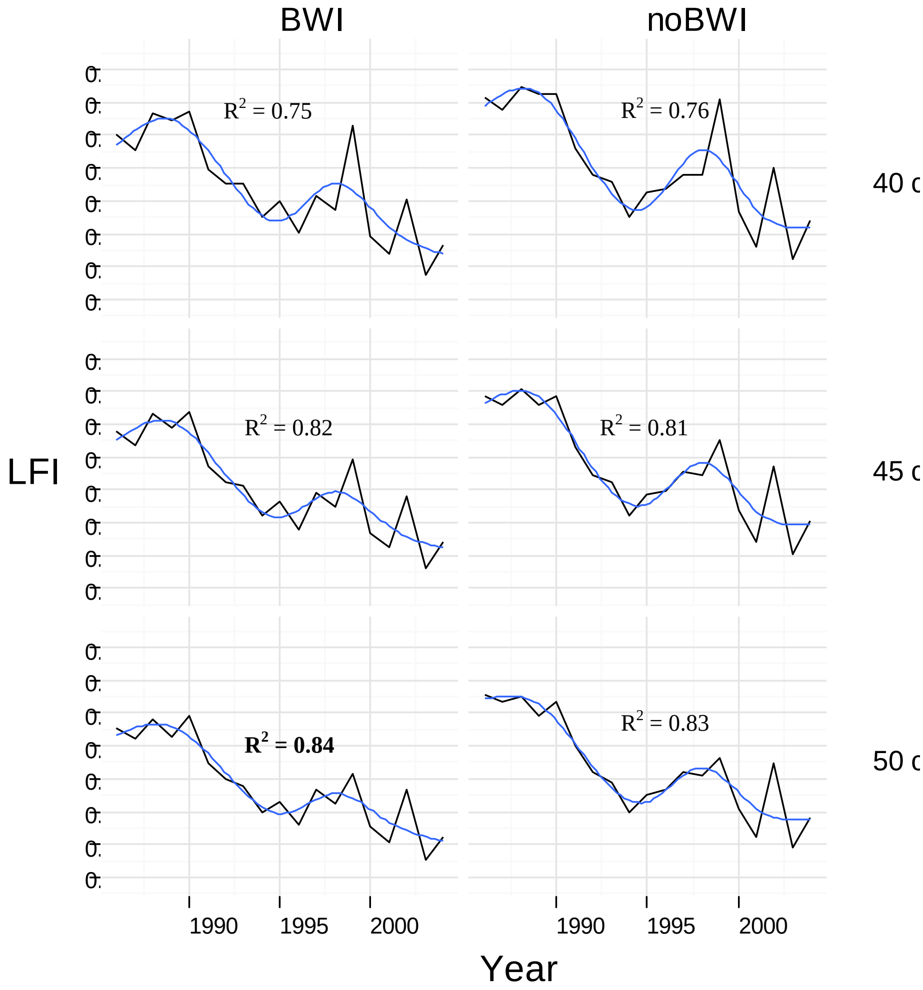

Figure 3.1.1. Time-series of the Celtic Sea Large Fish Indicator (LFI) for two species complexes: North Sea excluding boarfish (noBWI) and North Sea excluding boarfish and including blue whiting (BWI). Three length thresholds for large fish are used (40, 45 and 50 cm). The fitted line is a smoother (6th order polynomial) with fit (r^2) given for each line. The r^2 for the selected LFI (50 cm including blue whiting) is shown in bold.

Compared to the North Sea, the Celtic Sea community is characterized by the presence of larger fish, raising the question as to whether the 40 cm threshold was also the most appropriate threshold for the Celtic Sea. Shephard *et al*. (in review) examined this question explicitly, and using a similar signal-to-noise approach developed in the North Sea, established that a threshold of 50 cm was more appropriate (Figure 3.1.1); thus defining the Celtic Sea LFI as:

The proportion (by weight) of fish greater than 50 cm.

Unlike the North Sea, variation in the Celtic Sea LFI was primarily influenced by changes in the biomass of large fish, suggesting that the LFI was most affected in the Celtic Sea by the direct effects (the immediate removal of fish) by fishing.

Next, the Celtic Sea study considered what the most appropriate management target (EcoQO) should be for this newly defined LFI. Following the same approach adopted in the North Sea, trends in community-averaged fishing mortality were examined (Figure 3.1.2). This revealed an increasing trend in fishing pressure on the demersal fish community through the 1980s and into the 1990s. The LFI trend at this point suggested a major "tipping point" in 1990, after which the LFI started to decline markedly (Figure 3.1.2). On the basis of this, 1990 was considered to be the point when the community was in the least perturbed state. Consequently the LFI value in 1990 was chosen as the management target; an LFI of 0.4.

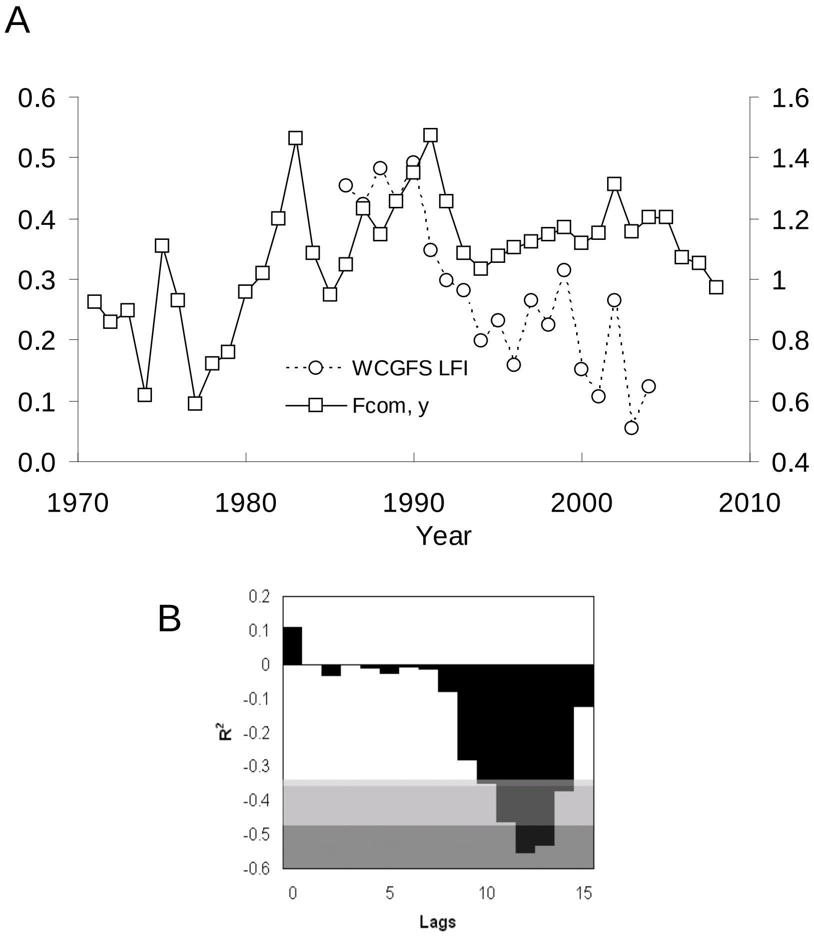

Figure 3.1.2. A: Trends in community-averaged fishing mortality ($\overline{F}_{com,y}$) and the LFI. B: Cross-correlations, expressed as r^2, between the LFI time-series and $\overline{F}_{com,y}$ at various time-lags (shading indicates significance levels: light grey 0.05>*p*>0.001; intermediate grey 0.001>*p*>0.0005; dark grey *p*<0.0005).

The Celtic Sea study demonstrates that the basic processes and principles underpinning development of the LFI for the North Sea "fish community" EcoQO pilot study are indeed transferable to other marine seas within the OSPAR area, and beyond, and illustrates the procedures that need to be taken to do this (Shepherd *et al*., in review). In regions where no single synoptic survey exists, these procedures may need to be applied independently to each of the various surveys required to achieve reasonable spatial coverage. The first steps in such an approach are illustrated in Section 1.3 below. The final step required in this process will be determining how the individual LFI trends derived from each survey should be integrated to achieve a regional assessment.

The LFI was initially considered to be the most useful indicator of the status of demersal fish communities because of its sensitivity to fishing disturbance (ICES 2001). Consequently, this relationship was explicitly examined in the North Sea and a 12–16 year lag in the LFI response to changes in fishing mortality was detected

(Greenstreet *et al*., 2011). The possibility of such a lag had been posited previously by Daan *et al*. (2005), who analysed the same IBTS data. Shepherd *et al*. examined the relationship between their newly defined LFI and fishing pressure in the Celtic and found significant relationships with a similar lag duration (Figure 3.1.2). The discovery of near identical lags in the relationship between fishing mortality and the LFI using this second, completely independent, dataset lends credibility to their reality (see also Section 1.2.2 below for lags in the Baltic Sea).

The presence of such long time-lags is of major potential significance from a management perspective. The implication is that it will take ten or more years for management measures implemented today to take effect. The LFI was chosen because of its supposed responsiveness to changes in pressure; such lag-times would seem to suggest otherwise. However, it must be pointed out that the LFI is an indicator of the status of the fish community. If the community response to exploitation involves lags of this duration, then such lags will be evident in any good state indicator. Both the North Sea and the Celtic Sea studies utilized the lagged regression models between F and the LFI to predict future behaviour of the LFI based on the fishing pressure trend over the previous ten or 15 years. Both studies suggested that measures to reduce fishing mortality taken over the last decade or so should bring about a recovery in the LFI. In the North Sea the measures taken may already be sufficient to achieve the EcoQO by 2020. The Celtic Sea data suggest that a further 20% reduction in fishing mortality, between 2009 and 2012, may be necessary to achieve the EcoQO by 2024 (Figure 3.1.3). These are key messages for management, but these simple statistical models are inadequate as the basis for firm advice. These results therefore underline the immediate need for process-based models to support scientific advice to underpin management towards LFI targets.

The survey on which the Celtic Sea LFI has been developed has been discontinued. An alternative survey, carried out at a different time of year and using a different gear, commenced a few years before this happened, but the period of overlap, when both surveys were undertaken, was too short to allow reliable inter-survey calibration. An LFI will need to be developed for this new survey and comparison of its trend with the current LFI's predicted trend may allow the two time-series to be linked so that an operational target could be set for the new LFI.

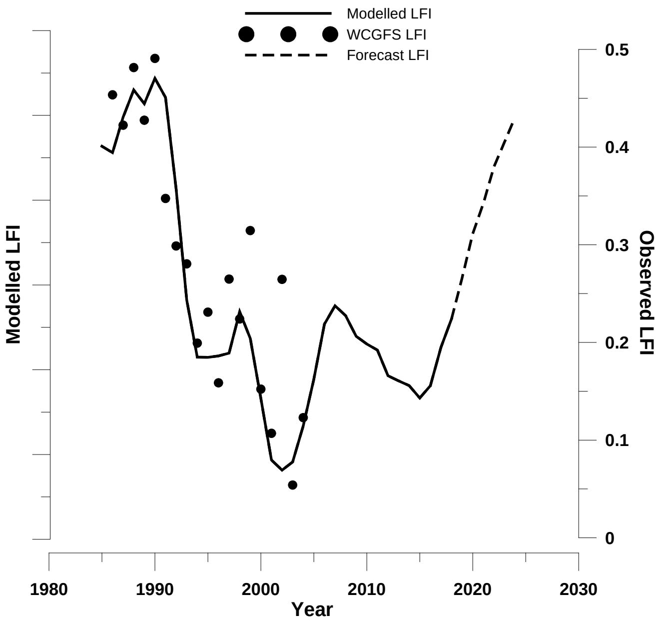

Figure. 3.1.3. Average modelled LFI predictions from five linear regression models regressing community-averaged fishing mortality on the WCGFS LFI with lags of 10y to 14y (following method in Greenstreet *et al.*, 2011). Since regression models "smooth" the data, the range of the predicted values was less than the range of the observed values. Hence the model predictions are plotted on a unit-less axis, which has been rescaled to match the observed data range. The trend line therefore shows "relative" variation in the LFI predicted by the model. Actual WCGFS LFI values are plotted to demonstrate the goodness-of-fit of the observed data to the model predictions.

3.1.1 References

Daan. N., Gislason, H., Pope, J. G., and Rice, J. C. 2005. Changes in the North Sea fish community: evidence of indirect effects of fishing. ICES Journal of Marine Science, 62: 177–188.

Greenstreet, S. P. R., Rogers, S. I., Rice, J. C., Piet, G. J., and Guirey, E. J. 2011. Development of the EcoQO for fish communities in the North Sea. ICES Journal of Marine Science, 68: 1–11.

Shephard, S., Greenstreet, S. P.R., and Reid, D.G. In Review. ICES Journal of Marine Science.

3.2 Examples of regional LFI series developed using protocol

In this section, the LFI protocol developed for the Celtic Sea is applied to survey datasets from the Baltic Sea and the international waters of the Grand Banks, NW Atlantic.

3.2.1 An LFI for the Baltic Sea

The Baltic Sea in northern Europe is one of the world's largest semi-enclosed bodies of brackish water. Biodiversity is very low and species are smaller compared to other regions. The "Large Fish Indicator" (LFI), developed for the North Sea, is one main indicators selected for the Marine Strategy Framework Directive (MSFD). Using the protocol described in Section 1.1, an LFI for the Baltic Sea is defined. This analysis was based on ICES DATRAS data from the Baltic International Trawl Survey, Quarter 1. Due to the salinity differences in the Baltic, a subarea of the Baltic Sea was considered, comprising Subdivisions 21, 22, 23, and 24. Temporal trends in biomass of the dominant species recorded in survey data were calculated. These were *Clupea harengus*, *Gadus morhua*, *Limanda limanda*, *Merlangius merlangus*, *Platichthys flesus* and *Sprattus sprattus*.

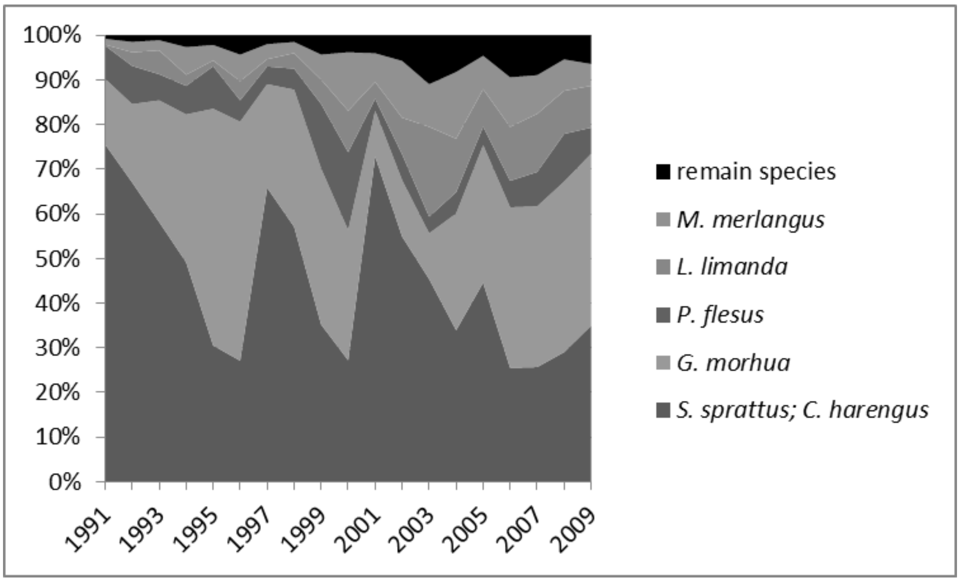

Figure 3.2.1.1. Ratio of biomass by species in Subdivision 21, 22, 23, 24 of the Baltic Sea over time (ICES, DATRAS, CatCatchWgt).

Since abundance in survey trawl samples was considered unlikely to be closely related to true abundance in the system, pelagic species *C. harengus* and *S. sprattus* were excluded from the analysis. For demersal species, *M. merlangus, L. limanda, P. flesus,* and *G. morhua* were selected, because these species comprise 80–95 % of total biomass in the survey-series (Figure 3.2.1.2).

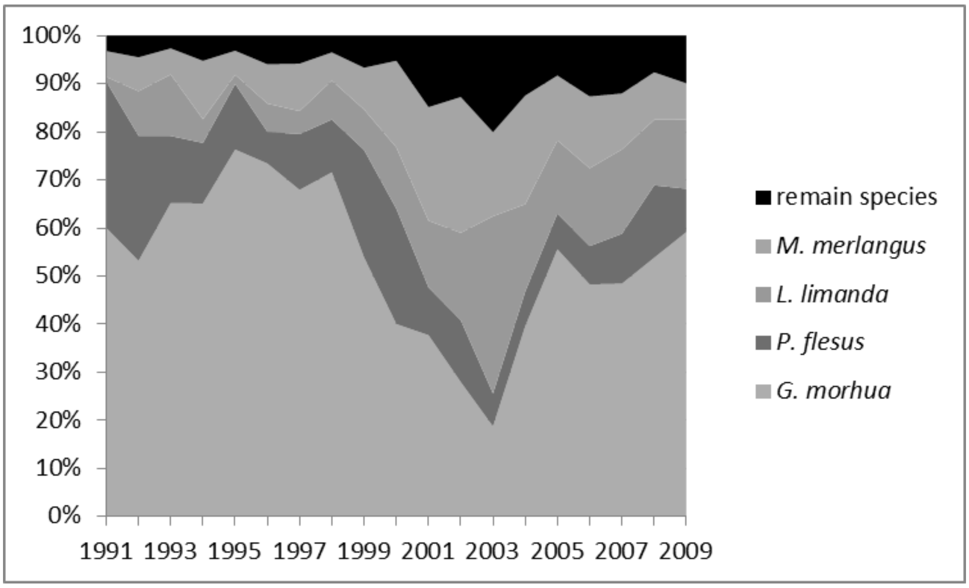

Figure 3.2.1.2. Ratio of biomass by species without *C. harengus* and *S. sprattus* in Subdivision 21, 22, 23, 24 of the Baltic Sea over time (ICES, DATRAS, CatCatchWgt).

To estimate the LFI, the length–weight relationship was calculated for each year for *G. morhua* and *P. flesus*. For *M. merlangus* and *L. limanda* the available single species data from 1991 to 2009 were summarized and length–weight relationships were calculated. Afterwards catch numbers at length were converted to weigh at length using the estimated weight at length relationships. After that, the biomass of individuals larger than 20 cm (LFI 20), larger than 30 cm (LFI 30) and larger than 40 cm (LFI 40) of each species and of all four species was divided by the total biomass of each species and all four species (Figure 3.2.1.3). The LFI based on a threshold length of 30 cm was considered to provide the most optimal signal-to-noise ratio. Because of lack of the required haul metadata, biomass was not standardized to biomass/km² as recommended in the Celtic Sea protocol. Figure 3.2.1.4 shows the total biomass of small and large fish over time for the four key species.

Data on fishing mortality F were available only for cod. To analyse the response of each LFI series to fishing mortality, the LFI and F were correlated at a sequence of lag periods. Significant correlations were observed at a time-lag of 14 yr (LFI 20; p = 0.023, rho: -0.523), (LFI 30: p = 0.014, rho: -0.560) and (LFI 40:p = 0.035, rho: -0.489). There were no significant correlations at shorter time-lags (2–12 yrs in 2-yr increments). This lag period corresponds to that found in both the North Sea and Celtic Sea situations.

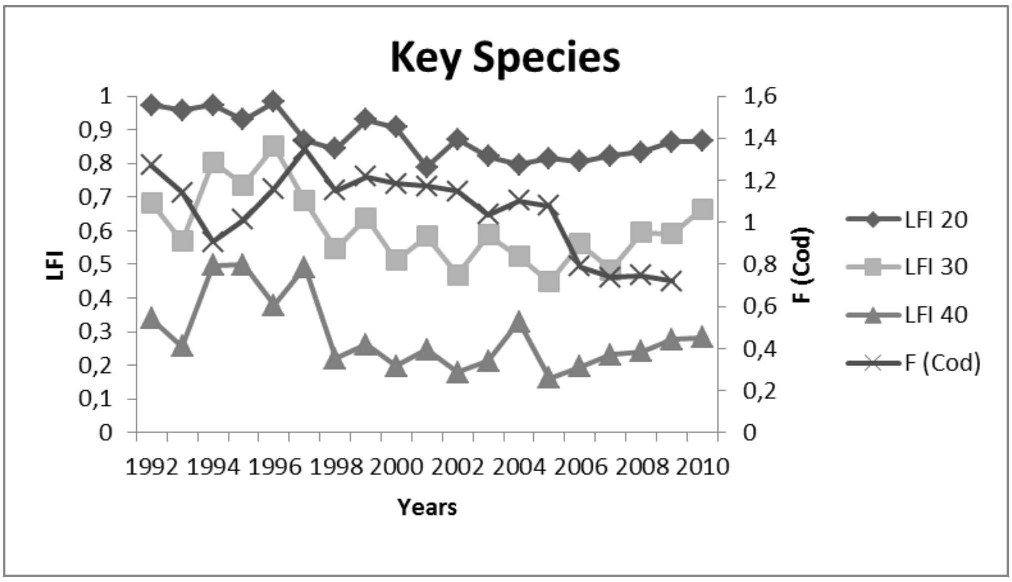

Figure 3.2.1.3. Estimated „Large Fish Indicator" for *P. flesus*, *L. limanda*, *G. morhua*, *M. merlangus*, and the four key species together in Subdivision 21, 22, 23, 24 (ICES, Datras, HLNoATLngt).

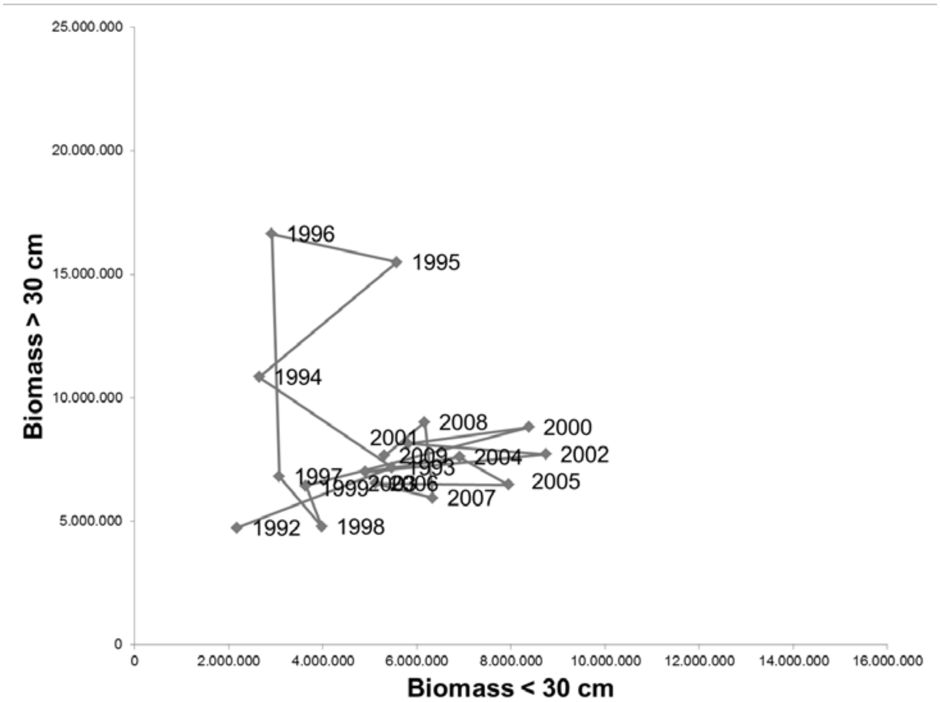

Figure 3.2.1.4. Plots of total biomass of small and large fish over time.

3.2.1.1 Conclusions

Due to the influence of brackish water, species diversity in the Baltic Sea is lower than in many marine systems, and the length of common fish species is smaller. Together these ecological differences demand some modification of the LFI as developed in the North Sea. Compared to other tested regions, the number of species in the selected species complex is very low, comprising only four species (*G. morhua, M. merlangus, P. flesus, L. limanda*). A length of LFI 30 cm represented the best tested large fish threshold. This is an exploratory analysis and further work is required.

3.2.2 An LFI for the Grand Banks of Newfoundland (international waters within NAFO Divisions 3N and 3O

The Grand Banks are part of the Canadian shelf, and comprise a series of submarine plateaus with depths ranging between 50 and 185 m. About 10% of their area lies outside the Canadian EEZ and spreads within NAFO areas 3N and 3O. The Grand Banks have supported an international fishery since 1400, but entered a period of marked decline during the 1970s, culminating in the collapse of the cod stock in the mid nineties and the closure of the major fisheries from 1995 (Olsen *et al*., 2005) until the present.

Fisheries statistics from NAFO show that there has been some fishing over the past 10 years. However, while Spanish effort represented 39% of the total in international waters, this fleet towed a maximum of only 20 times in most cells (0.05 degrees by 0.05 degrees) within the study area (Murillo *et al*., 2011; http://www.nafo.int/fisheries/frames/fishery.html).

Spain started surveying the international waters in the Grand Banks and Flemish Cap in 1995, on board a commercial trawler that used a “Pereira” type bottom trawl. When the research vessel Vizconde de Eza became available in 2001 the Pereira trawl was dropped in favour of the “Campelen” trawl used in the Canadian research surveys. Paired tows were carried out over several years to compare the performance of both gears, and in 2001 the paired tows served also to compare the performance of both ships and to obtain the coefficients necessary to standardize the older dataset. The whole survey covers international waters in NAFO areas 3N and 3O, Flemish Cap and 3L, which are organized as three consecutive surveys carried out from mid-late May to mid-late August. The maximum depth surveyed in the early years was about 700 m, but it was extended to 1600 m in 2002 (Figure 3.2.2.1). The goals of the survey in 3NO are to obtain biomass and abundance indices for the target species (*Hippoglossoides platessoides, Limanda ferruginea, Reinhardtius hippoglossoides, Glyptocephalus cynoglossus, Gadus morhua* and *Raja radiata*), to collect oceanographic data and, for in recent years, to collect information on invertebrates and vulnerable species. Biological information (length, weight, sex, maturity, stomach contents) is obtained from about 30 species, as well as otoliths from the target species.

The dataset used here is restricted to Divisions 3N and 3O and spans the period since the implementation of the Campelen trawl, 2002–2010. Due to time constraints it was decided to reduce the number of species in the analysis, which was restricted to the demersal species recorded in hauls taken at depths <=150 m. The analysis was performed for both the whole surveyed area and separately for hauls north and south of 44º N. The number of hauls per year used in the analysis was on average 13±2 (range: 9 (in 2010)–16) for the North area and 40±4 (range:31–47) for the South area.

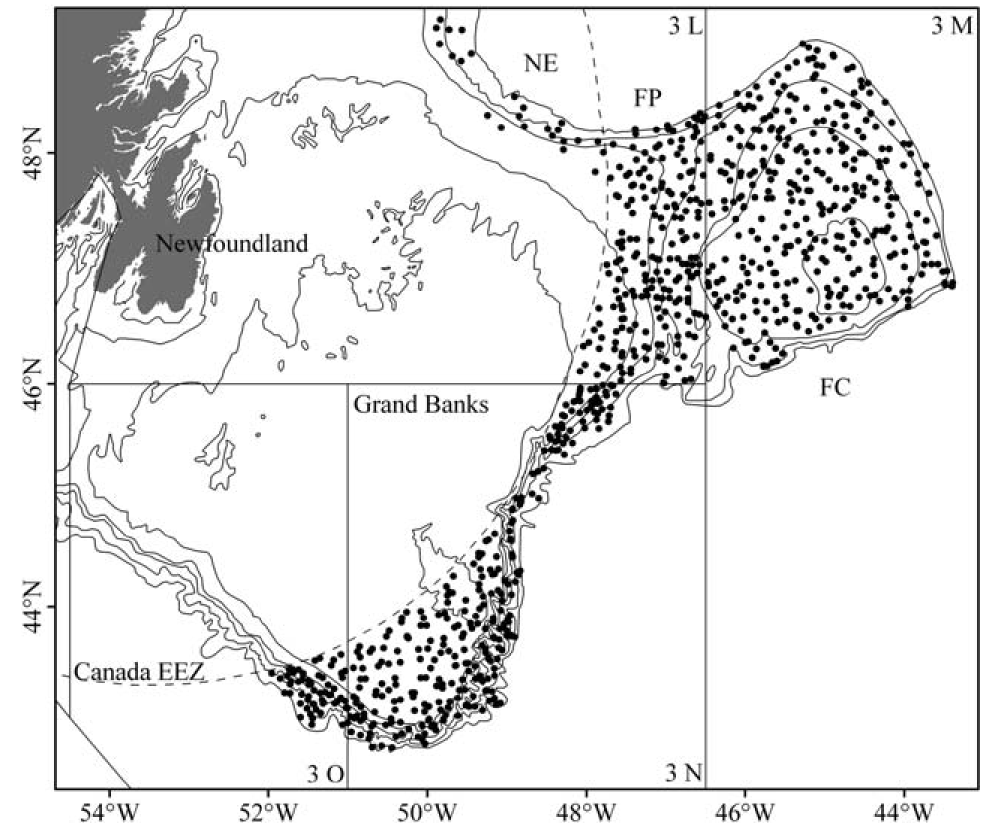

Figure 3.2.2.1. Map of the area covered by the Spanish survey. Dots show sampling locations. The depth contours correspond to depths 50, 100, 200, 500, 1000 and 150. FC, Flemish Cap, FP, Flemish Pass, NE, northeastern slope. From Murillo *et al*. (2011), with permission.

Catch and weight data were standardized after estimating the swept-area (km^2) per haul. The parameters α and β for the weight–length relationship were estimated from the survey data when possible, otherwise were obtained from fishbase.org. About 95% of the fish were smaller than 67 cm, and the threshold sizes selected to separate small and large fish were 45 cm, 65 cm and 75 cm.

Standardized weight of each species and for each length interval and haul were estimated for the calculation of the LFI index (biomass of large fish divided by total biomass) for the three different threshold sizes, 45 cm, 65 cm and 75 cm. The best smoother fit was found with a 6th order polynomial for threshold 45 cm, with r^2=0.8852 (Figure 3.2.2.2). A 55 cm threshold was also considered, but the LFI based on this threshold was out-performed by the 45 cm threshold LFI.

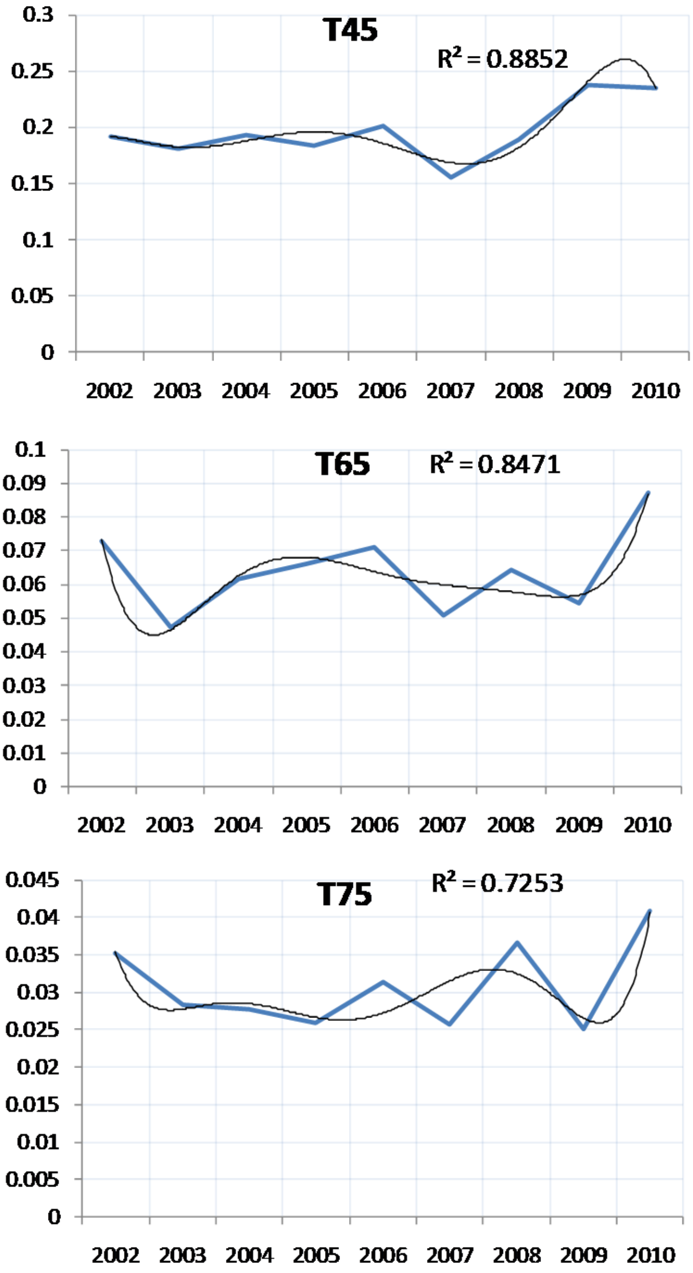

Fig 3.2.2.2. Time-series of the LFI for NAFO Divisions 3NO and the three thresholds selected (45, 65 and 75 cm). The estimated r^2 for each threshold area shown in the corresponding graphs.

Separated analysis of the data for the areas North (north of 44º N) and South (south of 44º N) showed very different trends of the LFI index (Figure 3.2.2.3). In the North area the index shows an oscillating trend that has decreased during the past three years, whereas in the South the LFI was very stable until 2007 and increased afterwards.

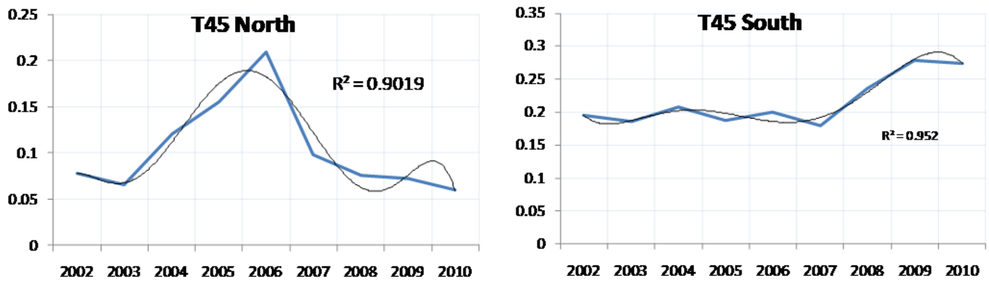

Figure 3.2.2.3. Time-series of the LFI (T45 cm) for the North and South areas surveyed in NAFO Divisions 3NO.

The state–space plots of relative biomass of small fish (≤45 cm) against biomass of large fish (>45 cm) showed that there were important changes in the demersal fish assemblage during the study period (Figure 3.2.2.4).

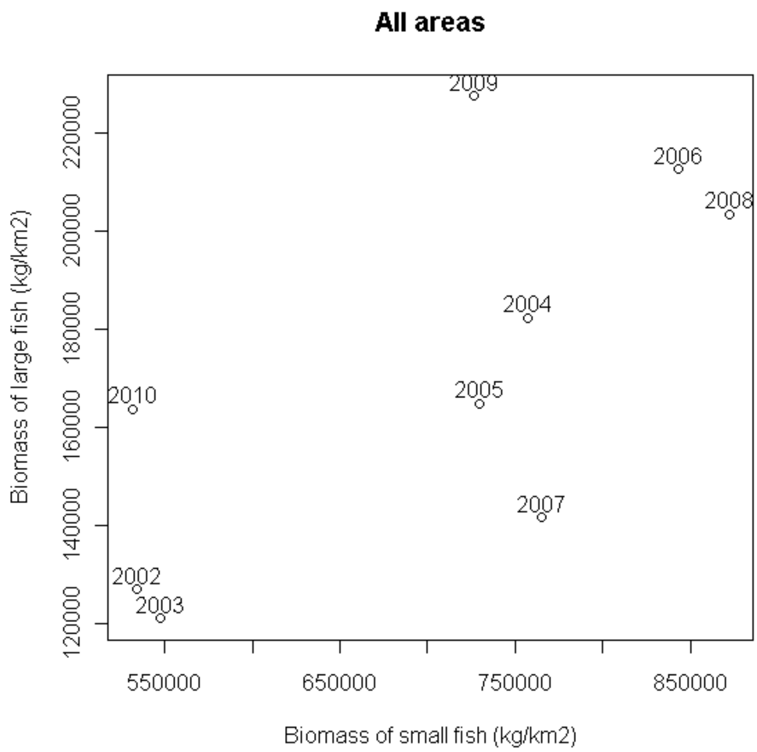

Figure 3.2.2.4. State–space plot for all the surveyed area in 3N and 3O.

These changes were also evident in the plots for the North and South areas, but the pattern was different than when considering both areas together (Figure 3.2.2.5).

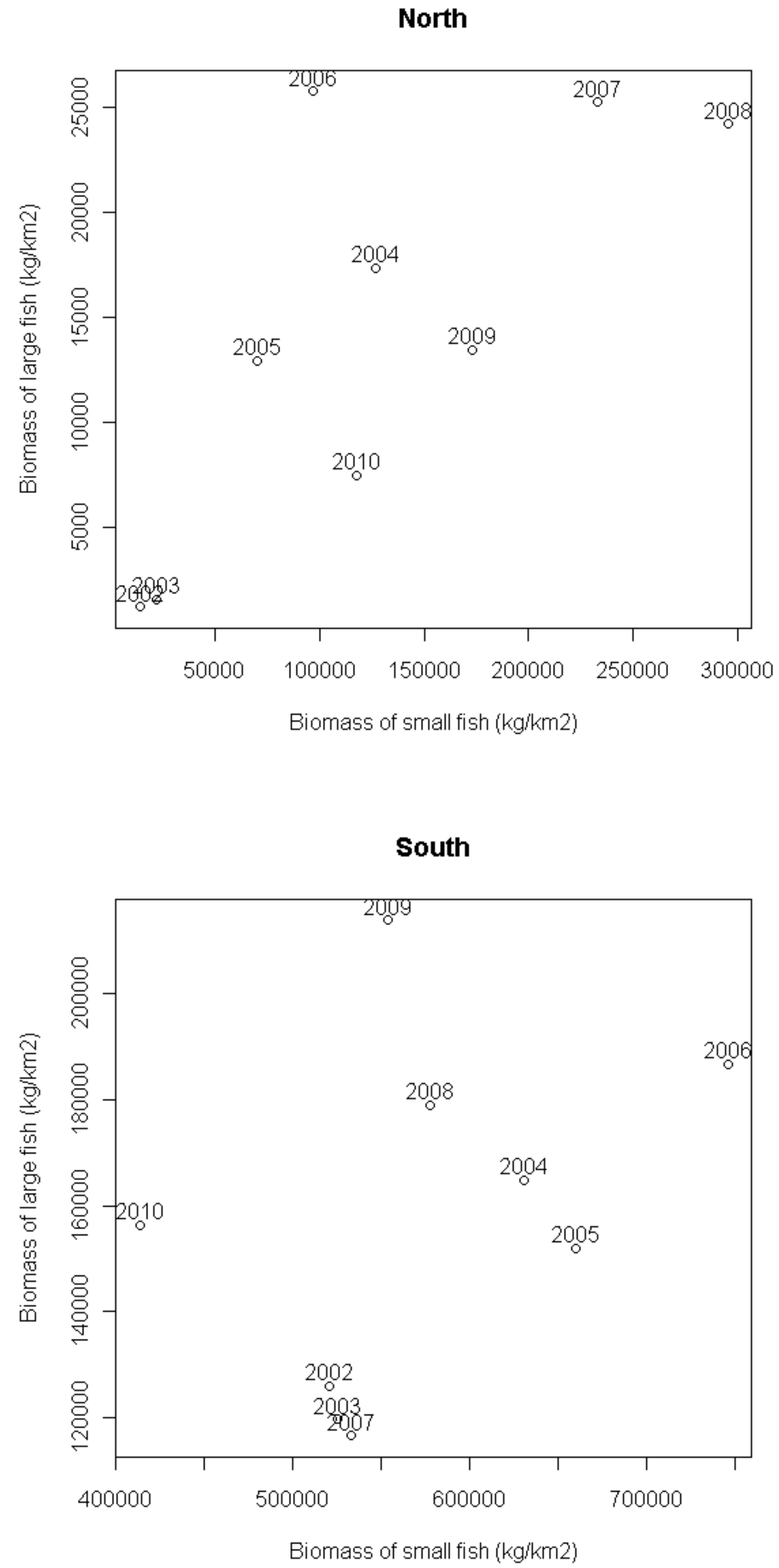

Figure 3.2.2.5. State–space plots for areas North and South.

These changes seem to be mostly driven by the oscillations in biomass of thorny skate (*Amblyraja radiata*), yellowtail flounder (*Limanda ferruginosa*), cod (*Gadus morhua*), northern sand lance (*Ammodytes dubius*), and American plaice (*Hippoglossoides plates-*

soides), as shown in Figure 3.2.2.6. As for the Baltic Sea, this is an exploratory analysis and further work is required.

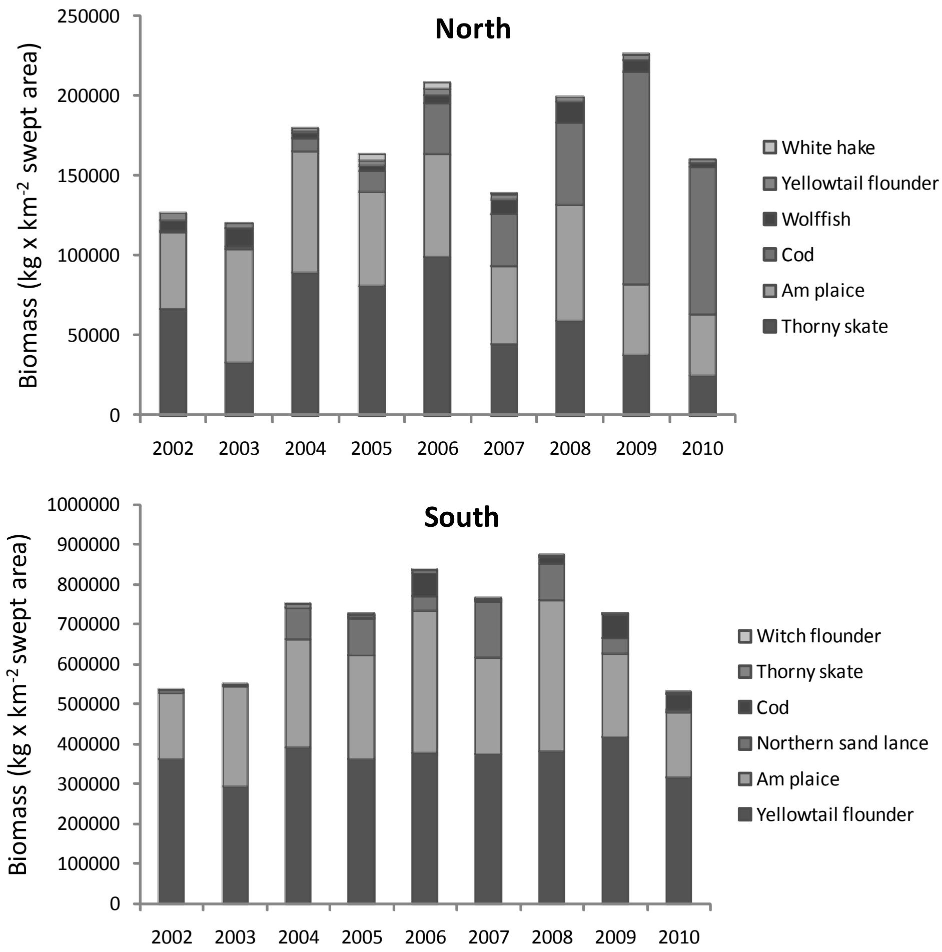

Figure 3.2.2.6. Oscillations in standardized biomass of the dominant demersal species.

3.2.3 References

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3.3 Developing a LFI for northern OSPAR Region III

All previous LFI analyses have been based on data from individual groundfish surveys. However, the North Sea is the only OSPAR region covered by a single survey. In this section, the first steps are made towards developing an OSPAR regional LFI based on the integration of LFI series from more than one survey.

The main groundfish surveys carried out in waters to the west of Scotland and around north coasts of Ireland are the first and fourth quarter Scottish West Coast Surveys (Q1SWCS and Q4SWCS) respectively. The whole area covered by these surveys was partitioned into two regions: the "inner" region consisting of the Minches and the more coastal waters down into the Irish Sea; and the "outer" region consisting of the Hebridean continental shelf waters and the continental shelf waters to the northwest of Ireland (Figure 3.3.1). Table 3.3.1 which gives sampling effort by each survey in each region and year, suggests that sampling effort was generally adequate to generate a reliable LFI.

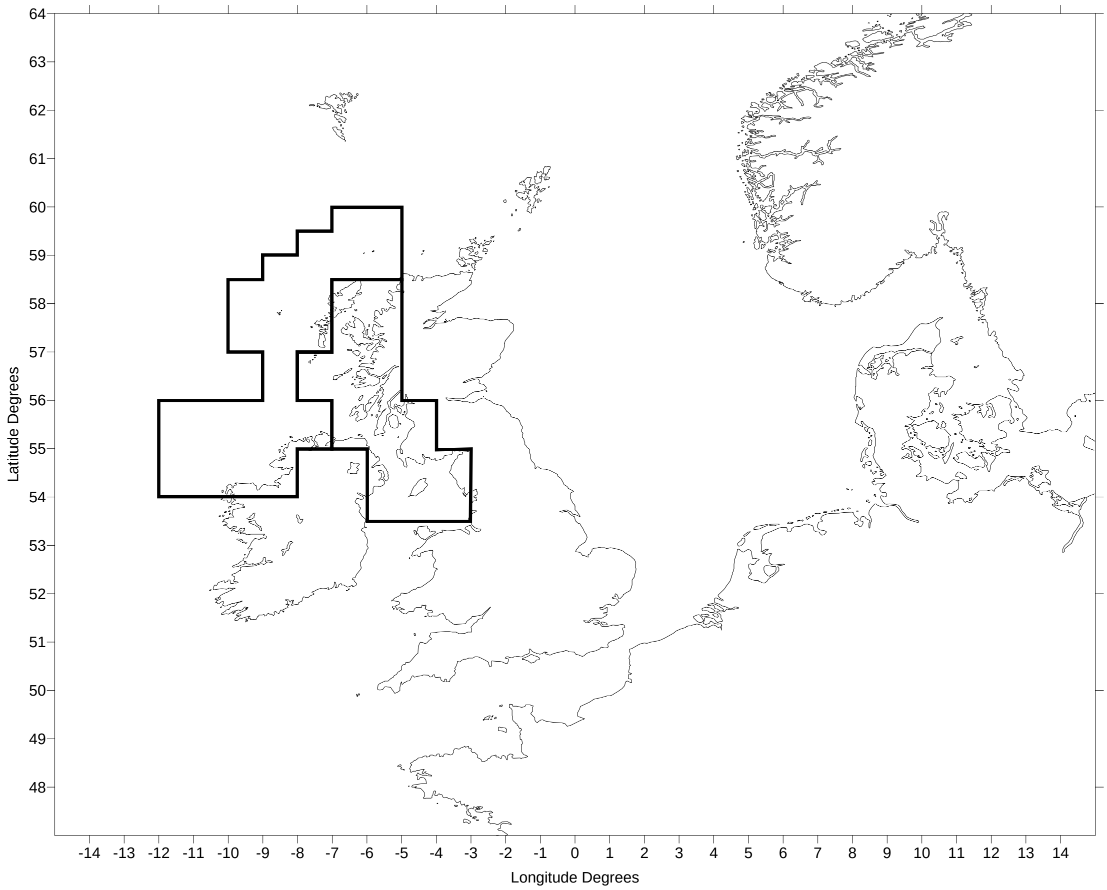

Figure 3.3.1. Chart showing the locations of the "inner" and "outer" regions for which Scottish West Coast Groundfish Survey (SWCGS) data were analysed.

Table 3.3.1. Sampling effort information: numbers of hauls and area swept (km-2) in each region and quarter (Q1 is first quarter and Q4 is fourth quarter).

Year	Inner Region				Outer Region			
	Q1	Q4	Q1	Q4	Q1	Q4	Q1	Q4
	Number		Swept Area		Number		Swept Area	
1986	14		1.9805		15		2.1873	
1987	17		2.5488		23		3.6907	
1988	19		2.8883		20		2.8102	
1989	18		2.9454		19		3.5109	
1990	18		2.8839		16		2.5467	
1991	20		3.3816		23		3.9619	
1992	17		2.5666		18		2.5202	
1993	17		2.4803		18		1.8479	
1994	18		2.4750		19		2.7993	
1995	16		2.3484		20		2.4314	
1996	25		3.3471		20	31	2.4909	3.8307
1997	31	25	2.9534	2.8448	20	29	2.4405	3.9950
1998	31		2.7328		18		1.6610	
1999	38	23	2.4762	1.4330	20	27	1.3269	1.7874
2000	36	28	2.3332	1.7880	23	35	1.6014	2.3169
2001	31	28	2.1418	1.9279	20	39	1.5509	2.8484
2002	32	29	2.0703	1.9045	21	46	1.4188	3.0321
2003	34	30	2.0465	2.0341	32	43	2.1241	2.9548
2004	32	30	2.0794	1.9236	28	41	1.9839	2.7818
2005	31	30	1.9358	1.9245	27	46	1.7844	2.8492
2006	31	11	1.8969	0.6927	30	46	1.8018	2.7750
2007	18	20	1.2050	1.2756	28	52	1.5465	3.2504
2008	18		1.1249		31		1.8456	

The LFI was defined as the proportion of total biomass exceeding a threshold length. Following established practice (Greenstreet *et al.*, 2011; Shephard *et al.*, in review) several thresholds were examined to determine which provided the best signal to noise ratio (best fitted by a 5th order polynomial function). Likewise consideration was given as to which species should be included in the demersal species assemblage. On the basis of these tests the optimal threshold length was deemed to be 45 cm and the optimal species suite excluded blue whiting (included in the Celtic Sea LFI but excluded in the North Sea LFI) and also excluded boar fish (excluded in the Celtic Sea LFI but included in the North Sea LFI). Trends in the LFI in both regions suggested a marked deterioration in the status of the demersal fish community during the early 1990s, followed by a prolonged period in a relatively stable "perturbed" condition, with some indication of a recovery since 2005 (Figure 3.3.2). In both regions, the two LFI trends were significantly correlated ("inner" r^2=0.97; "outer" r^2=0.60) over the period when the two surveys overlapped. Either could therefore be used to monitor change in the LFI, but the Q1 survey hold advantages over the Q3. Firstly the Q1 survey is the longer time-series of the two, so would be the more useful in terms of setting a target as the EcoQO. Secondly the timing of the survey coincides with the survey used to generate the North Sea LFI, so perhaps provides a better comparison of changes in the status of the demersal fish community in neighbouring regions.

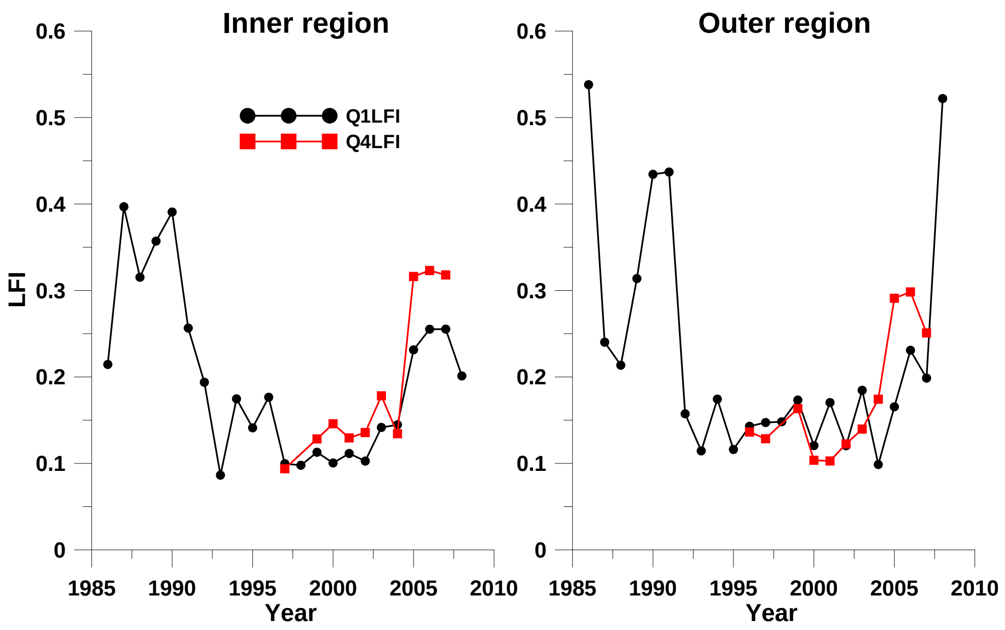

Figure 3.3.2. Trends in the LFI derived from the Q1 and Q4 SWCS determined for both the inner and outer regions of northern OSPAR Region III.

While remaining beyond the scope of this meeting, the next stage of this analysis would be the development of a protocol for integrating data from separate surveys, when this is necessary to obtain coverage of an entire OSPAR region. The Celtic Sea study reported in Section 1.1 covers the southern part of OSPAR Region III, while the data presented here cover the northern part. The next step therefore is to develop the protocol for integrating the assessments undertake in both the northern and southern parts of the region to derive an overall assessment of the status of the demersal fish community across the entirety of OSPAR Region III. Further work is therefore required to build on these initial analyses.

3.4 Recovery in the LFI after reductions in fishing mortality

As discussed above, management advice based on the LFI is likely to require the application of process-based ecosystem models. In this section, WGECO briefly review available models (but see ICES, 2010) and describe the use of a new model in predicting likely recovery trajectories in the LFI following changes in fishing mortality.

In the last WGECO report (ICES, 2010), a multispecies size-structured model of the North Sea, developed by Strathclyde University and Marine Scotland, was presented. The basic structure of this model, set up to provide advice to underpin the cod stock recovery programme, has now been published in Speirs *et al*. (2010). Explicit application of the model to support LFI advice is still in final preparation, but its application to assess LFI recovery rates for the North Sea was detailed in the previous report (ICES, 2010). However, as noted in the conclusions of that report (ICES, 2010), results for the Speirs model should be compared with simulation results from other models, to ensure that management advice is based on a broad scientific knowledge base. Importantly, such a comparison also helps to test the sensitivity of results to model structure.

Thus, we describe the development of two new models and their application in assessing LFI response to and recovery from fishing. The first is the Population-Dynamical Matching Model (PDMM) and the second is the Fish Community Size-

Resolved Model (FCSRM). These models differ from the Speirs model by being less specific to a particular subregion such as the North Sea; they are more generic. Nonetheless, the model communities exhibit general features of real temperate marine communities and can thus be used to examine general underlying trends. Results from these models may also be useful for informing management in the absence of sufficient empirical data for parameterization of the Speirs model (2010).

3.4.1 Population-Dynamical Matching Model (PDMM)

The first model is the Population-Dynamical Matching Model (PDMM). It is a multispecies size-structured model that uses a community assembly algorithm to generate model communities with thousands of dynamically coexisting and interacting species, ranging from phytoplankton to large fish. Model species are characterized by different maturation body sizes and a set of species-specific traits. Together, these traits determine population dynamics and the community foodweb. The PDMM was parameterized for a temperate shelf community in the Northeast Atlantic, and includes 189 fish species. A more detailed summary of the PDMM model structure is given in Box 1.4.1.1; full details of the structure and performance of the PDMM in relation to LFI can be found in Rossberg *et al.* (2008) and Fung *et al.* (in review).

PDMM MODEL STRUCTURE

The PDMM distinguishes producer and consumer species. Apart from the time-dependent biomasses of all species (Rossberg and Farnsworth, 2010), each producer species is characterised by a maturation body mass, five abstract competition traits and five abstract vulnerability traits. Consumer species are characterised by a maturation body mass, five abstract foraging traits and five abstract vulnerability traits. The abstract traits specify points in a five-dimensional competition or trophic niche space. The strength of a trophic interaction between a consumer and its resources (producers or consumers) is larger the closer the consumer's foraging traits are to the resources' vulnerability traits in niche space (Rossberg et al. 2010a,b,c), and the closer the predator-prey size ratio is to a preferred value. Thus, body sizes together with the foraging and vulnerability traits specify the foodweb. In addition, producers compete if they have similar competition traits. Body size also partly determines population growth and turnover rates (Peters, 1983).

The trait values of all species are determined through an iterative assembly algorithm: Starting from a community with a few species, new species are generated by modifying the traits of extant species at random (thus generating phylogenetically structured foodwebs; Bersier and Kehrli, 2008) and added to the community if they can invade it; species are removed from the community if they go extinct. In this way, the foodweb is gradually built up. This method to generate complex communities overcomes a problem of marine foodweb modelling first described by Andersen and Ursin (1977, Section 2.4): Models of speciose, empirically parameterized foodwebs tend to become dynamically unstable unless sufficiently strong non-trophic intraspecific competition is incorporated (e.g., Andersen and Ursin, 1977; Loeuille and Loreau, 2005; Andersen and Pedersen, 2010; Harvig et al., 2011). The assembly algorithm of the PDMM leads to stable communities in which competition among consumer species arises only from well-understood trophic mechanisms (resource- and consumer-mediated competition).

Box 1.4.1.1. A summary of the PDMM model structure.

In total, 30 exploratory fishing scenarios were examined by Fung *et al*. (in review). These regimes differed according to (i) the size-range fished, (ii) the fishing intensity and (iii) the number of years of fishing. During each fishing scenario, the LFI dynamics over time were tracked. An LFI threshold of 40 cm was used (as for the North Sea; Greenstreet *et al*., 2011). For each fishing scenario, the LFI at the end of the fishing period was recorded to examine the extent of changes in LFI under the different fishing variables considered. After application of each scenario, following WGECO (2009), the LFI dynamics in the hypothetical absence of fishing were tracked, to determine the equilibrium LFI after recovery and the time to approach this value.

Interestingly, for all cases considered, the LFI decline was similar for fishing durations of 25 and 50 yrs (Figure 3.4.1.1). This shows that with sustained fishing, the LFI came close to a new equilibrium in <25 yrs. However, fishing for longer can drastically reduce the LFI equilibrium reached in the recovery phase (Figure 1.4.1.1). This result reflects local population extinctions of large model fish species. In real fish communities, there is a theoretical and empirical basis for such localized extinctions (Dulvy *et al*., 2003), particularly for vulnerable large-bodied species with low growth rates. In addition, the time taken for the LFI to recover to near equilibrium was typically on the order of decades, regardless of the fishing scenario applied. In addition, it was found that the recovery trajectories asymptoted following exponential functions (Figure 3.4.1.2), consistent with results from the Speirs model, reported in ICES (2010).

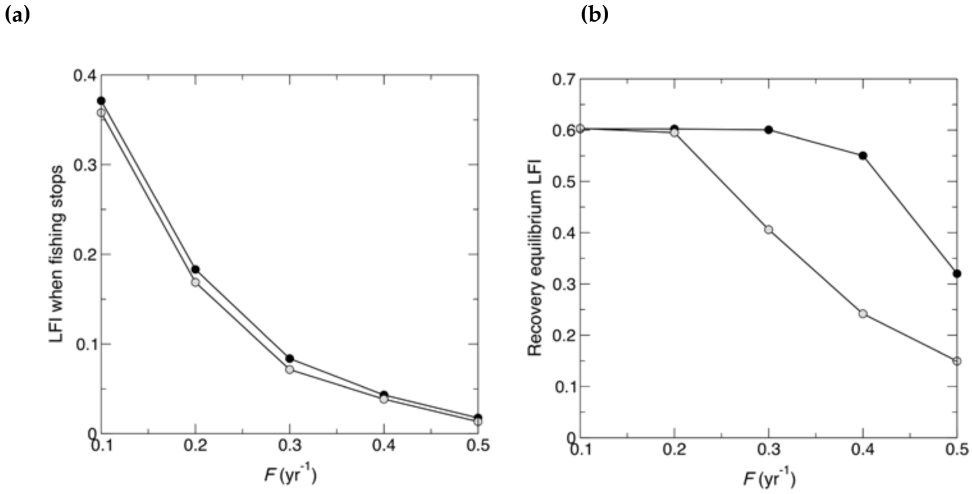

Figure 3.4.1.1. LFI against *F* (a) after non-selective fishing for 25 yrs (black circles) or 50 yrs (grey circles) and (b) after subsequently letting LFI relax to unexploited equilibrium. The solid lines are for visual guidance. Based on the PDMM.

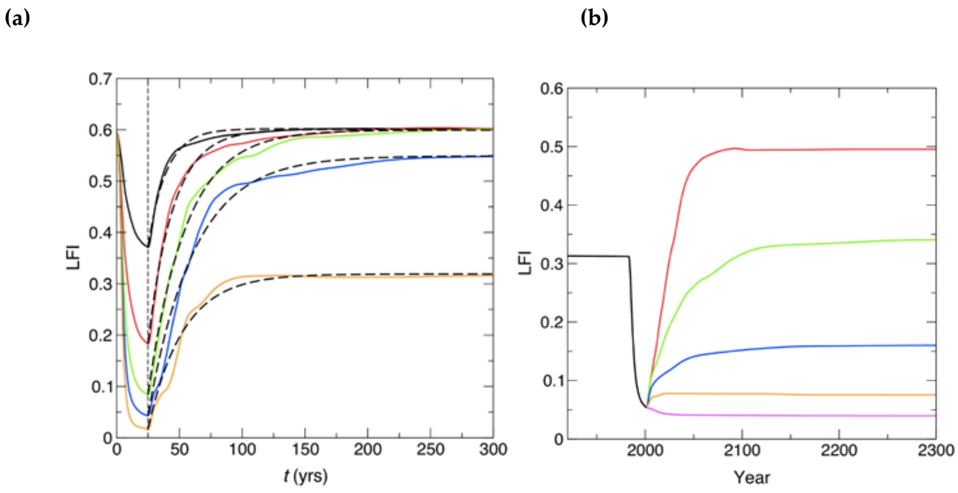

Figure 3.4.1.2. LFI decline and recovery for (a) 25 yrs of non-selective fishing of a pristine model community at *F* = 0.1 (solid black line), 0.2 (red line), 0.3 (green line), 0.4 (blue line) and 0.5 (orange line) yr^{-1}, and (b) for a fishing scenario resembling that of the North Sea. For (a), the dotted black lines are exponential fits for the recovery LFI trajectories. Based on the PDMM.

The PDMM community was also used to examine possible recovery rates for the North Sea. Although the model was not specifically parameterized for the North Sea, it has general characteristics of a temperate shelf community in the Northeast Atlantic, of which the community in the North Sea is a subset. Thus, model results may reflect general trends underlying North Sea fish community dynamics. For this investigation, a fishing scenario was used that produced LFI decline resembling that observed for the North Sea between 1920–2001 (Greenstreet *et al.*, 2011). The rates of recovery to the proposed reference value of 0.3 (Greenstreet *et al.*, 2011) were then calculated for different values of *F*. It was found that recovery back to the baseline was only observed for $F < 0.2$ yr^{-1}, with times of 24 and 82 yrs for $F = 0$ and 0.1 yr^{-1} respectively. Again, the recovery trajectories asymptoted following exponential functions.

3.4.1.1 Conclusions

The finding that model LFI recovery to near equilibrium typically takes multiple decades suggests that in general, recovery plans for fish community structure, as measured by the LFI, need to be implemented on decadal or longer time-scales. The model recovery trajectories, following exponential functions, suggest that LFI recovery could initially be quick, but then slow down.

3.4.2 Fish Community Size-Resolved Model (FCSRM)

This section describes the FCSRM and summarizes LFI results from Houle *et al.* (in review), in which further details can be found.

The FCSRM is a multispecies size-structured model based on the model by Hartvig *et al.* (2011). Unlike the PDMM, as well as modelling different fish species, it models the dynamics of intraspecific population structure for each species. For each fish species, the FCSRM models the processes of reproduction, growth, metabolism and prey encounter at an individual level. The model was parameterized for a generic temperate fish and planktonic community and used to assess the sensitivity and specificity of the LFI to fishing.

A logistic-type selectivity (Millar and Fryer, 1999) was used for the trawlnet, with a mesh size of 10 cm. This gave a length at 50% selectivity of 31 cm, the average for cod, haddock, sole and whiting sampled from the North Sea (Piet *et al.*, 2009). A Gaussian selectivity curve (Millar and Fryer, 1999) was used for the gillnet, with a mesh size of 120 mm, following data from Irish/Cornish hake gillnets (Revill *et al.*, 2007). The fishing mortality at a specific body size is given by a constant fishing effort E (measured in yr^{-1}) multiplied by the selectivity.

In addition, to investigate the effects of ecological and environmental variability unrelated to fishing, for each fishing scenario and each community structure, 100 simulations were performed using 100 randomly varied parameter sets.

The LFI showed a strong negative trend with effort E for trawl fishing, but only a weak negative trend for gillnet fishing (Figure 3.4.2.1). For all cases, the LFI variation due to environmental and ecological variability was small (Figure 3.4.2.3.1), especially compared with other indices for community size/trophic structure (see Houle *et al.*, in review).

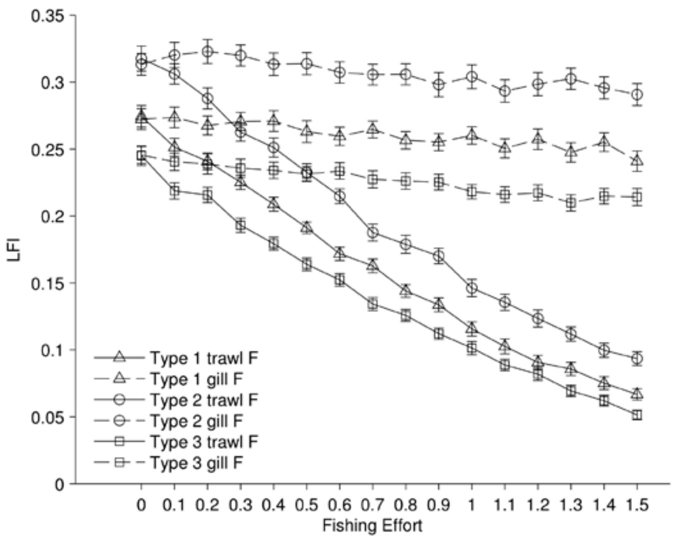

Figure 3.4.2.1. For the FCSRM, LFI against fishing effort *E* for trawl fishing and gillnet fishing, and three fish community types. For each point, the standard deviation over 100 simulations with 100 random parameter sets is shown as a vertical bar. Types 1 to 3 communities differ by the distribution of species over sizes.

These model results suggest that the LFI is very sensitive and consistently responds to non-selective trawl fishing, specifically compared with the effects of variations in environmental and ecological conditions. This result supports the broad use of the LFI in different geographic regions for detecting the effects of trawl fishing on fish community structure. However, the results suggest that the LFI does not do so well in detecting the effects of targeted fishing. Empirical verifications of these conclusions are highly desirable.

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3.5 Comments on the use of the LFI as a 'foodweb' indicator for the Marine Strategy Framework Directive

Last year's WGECO report (ICES, 2010) stated that:

"During the ICES/JRC process to identify indicators for the descriptors of good environmental status under the MSFD, the LFI has recently been suggested as one indicator of 'foodwebs'. The Commission decision (as required under Article 9(3) states, under the heading 'Structure of foodwebs (size and abundance)' a criteria 'Proportion of selected species at the top of foodwebs', as follows;

The rate of change in abundance of functionally important species will highlight important changes in foodweb structure. Indicators are to be developed for large fish (by weight) (4.5), using the experience in some subregions (e.g. North Sea). For large fish, data can be used from fish monitoring surveys, on an annual basis, at the scale of a regional or subregional sea.

The LFI has also been used in the UK as one indicator to describe 'marine ecosystem integrity' (http://www.jncc.gov.uk/page 4229), and also as a supporting indicator for the UK governments' Natural Environment Public Service Agreement, to monitor progress towards achieving the vision for *clean, healthy safe, productive and biologically diverse oceans and seas*."

However, since the LFI was not developed as a foodweb indicator, its utility in this role has not been fully investigated. WGECO now highlights this issue and suggests that further research into indicator function would be valuable before the first review of the MSFD in 2018. Studies using process-based models, as described above, are likely to be useful as they have the capacity to output both changes in the rates of energy flow between different components as well as potential changes in energy flow pathway.

Measures of the average trophic level of landings, or the system, have received much attention based on the theory that fishing leads to a reduction in trophic level (Pauly *et al.*, 1998). However, more recent studies have found that trophic level does not always track fishing pressure (Piet and Jennings, 2005; Branch *et al.*, 2010), and the average trophic level of landings responds not only to ecosystem status but also to fishing patterns (Essington *et al.*, 2006), which confounds interpretation of landings-based indicators of foodweb status.

Measures of community size structure have been proposed (e.g. Kerr and Dickie, 2001) as an alternative framework to provide robust indicators of the effects of fishing on the fundamental trophic structure of marine ecosystems. This is due to the fact that predator–prey relationships in aquatic environments are strongly size dependent and that fishing is size selective and can lead to a reduction in the average size of the fish community (Bianchi *et al.*, 2000). These observations suggest that fisheries-induced changes in size structure are associated with changes in trophic structure. This is well supported by macroecological theory. Comparative studies of the ability of different indicators to show fishing signals have demonstrated that size-based indicators are responsive to the effects of fishing (Bianchi *et al.*, 2000; Jennings *et al.*, 2002; Greenstreet and Rogers, 2006), even in the presence of confounding drivers (Blanchard *et al.*, 2005; Houle *et al.*, in review).

However, studies that have examined both the change in size composition and trophic level at which predators are feeding suggest contradictory evidence. In the northwestern North Sea, in a region heavily affected by fishing activity, the anticipated change in demersal fish size composition was observed, However, no change in average trophic level at which the demersal community was feeding was detected; the same proportion of piscivores was present, but they were smaller in length; large piscivorous individuals were replaced by small ones (Jennings *et al*., 2000). Such observations call into question the role of size-based indicators as indicators of change in marine foodwebs. The role of such indicators, the LFI for example, should be explored in this respect before too much reliance is placed on such indicators in determining foodweb GES for the MSFD.

3.5.1 References

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3.6 Conclusions

In ToR a, WGECO addressed the need to establish a consistent process for applying the Large Fish Indicator (LFI) to marine regions outside the North Sea. This required both a technical protocol and additional research into the properties of the LFI as a management tool. Key issues identified while developing the generic LFI protocol were the need to define an appropriate region-specific fish species complex and a corresponding "large fish" length threshold. In particular, the chosen complex must be robustly sampled by the survey gear, and should typically include species that represent key functional roles in the demersal fish community.

LFI analyses for the North Sea (ICES, 2010), Celtic Sea and Baltic Sea all identified lags of approximately 10–15 y in the response of the LFI to changes in fishing mortality. The discovery of such a lag in these fairly disparate marine systems suggests that the lag may be a generic phenomenon that must be accommodated when using the indicator for management advice. Further investigation of this apparent lag is therefore desirable. Process-based models will likely be useful for identifying and understanding the processes underlying such lags and offering meaningful predictions of future states. The theoretical model results appear to infer an almost immediate response in the LFI following reductions in fishing effort, and this would appear to contradict the lags observed in the majority of the empirical studies. This apparent discrepancy needs investigation, but it is important to realize that severe reductions in F in a model environment will inevitably bring about an increase in the LFI because fish growth is a constant process; growth causes the biomass of large fish to increase at all times. In reality however, only if F is reduced to the point that biomass growth in large fish exceeds the biomass loss associated with fishing mortality will a similar immediate response in F be observed. In situations of greater mortality, the recovery of the LFI will rely on the occurrence of larger than usual cohorts of fish growing to exceed the threshold length, and this would be dependent on the stochastic nature of recruitment process. Following ICES (2010), some discussion of available models and their use is provided. While outputs from these models suggest recovery periods of several decades, recovery trajectories may take an exponential form, implying that initial recovery rates will be high so that large improvements may occur over short time-scales, although full recovery to a new "equilibrium" state may take considerably longer.

Finally, WGECO comments on the use of the LFI as a "foodweb" indicator within the MSFD. It is concluded that the LFI, as a size-based metric, may function well as an indicator of marine foodweb structure. However, WGECO suggests that further research is required so that this can be clarified before the first review of the MSFD in 2018.

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4 ToR B: Integrated ecosystem management plans

b) Review the use of science in the development and implementation of “integrated ecosystem management plans” (IEMPs) including objectives setting and performance evaluation as well as other considerations.

The second ToR was planned to address Integrated Ecosystem Assessment (IEA; also discussed in 2010), but it was decided to defer this ToR for one year pending reports from IEA Expert groups (WGNARS, WGIAB, WGINOSE and WGEAWESS).

5 Tor C: Review and comment on the SGMPAN report which presents general guidelines for MPA network design processes that anticipate the effects of climate change on marine ecosystems

5.1 Preamble

The 2011 report (ICES 2011) of the Study Group on Marine Protected Area Networks (SGMPAN) considered how MPA networks should be designed to account for changes resulting from climate change. While WGECO did not feel qualified to provide a detailed review of the report, it was felt that the report was clearly written, authoritative and provided a robust evidence based account of the current issues as they relate to the east coast of North America. However, WGECO have previously noted that while MPAs are an powerful and appropriate management tool to achieve some objectives they are not the solution to many of the challenges that will need to be met to achieve sustainable ocean management (ICES 2004).

WGECO are of the opinion that any MPA network designed around current conditions may not be fit for purpose in the face of continued climate change. Given the depth and quality of the SGMPAN report WGECO decided to:

1) Focus on the theoretical framework discussed;
2) Consider what material would need to be included to make the report relevant to the NE Atlantic and hence the entire ICES area;
3) Link the report to the predicted climatic impacts expected in the OSPAR area developed by ICES (Tasker, 2008) in support of the QSR;
4) Consider how the available evidence might be used to develop practical and pragmatic guidance on MPA network implementation.

5.2 Issues associated with the theoretical framework

WGECO reviewed the ecological framework material produced by SGMPAN (Chapter 3 of their report) and it stimulated considerable debate. With the benefit of a different set of experts, a different range of expertise and experience and the temporal separation for the original work a number of new perspectives emerged which we believe compliment and strengthen the work and should be considered by SGMPAN when they next meet (by correspondence) in June 2011.

5.2.1 Management objectives requiring MPAs

WGECO felt that the report did not explicitly consider the drivers for MPAs and MPA networks. As previously noted by WGECO when discussing MPAs as management tools (ICES 2004), MPAs are commonly designated to achieve one of two types of objective. Either the protection of representative areas with their associated habitats, species and hence ecological functioning or the protection of specific habitat features or species for example species of particular conservation concern. WGECO considers it critical that the objectives of the MPA network are considered alongside the implications of climate change.

5.2.1.1 Representative MPAs

The latest stage in the application of the CBD sees a commitment to "10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes." It would seem that if such a network is in place then it should essentially be robust in the face of climate change, with caveats described below.

Our thought experiment runs as follows. An area of subtidal mud at 50°N is declared a MPA as a 'representative area' for that habitat type. An area at 55°N is also designated as a representative MPA. The two areas each contain different biological assemblages consistent with their different latitudes. We further assume that other, similar, areas at each latitude form a local MPA network that support each other. Time passes and the area warms causing a change in environmental conditions. We can envisage a replacement of the species at the 55°N site by those previously at 50°N, which has become too warm for them. Extending this then the site at 50°N would now be occupied by assemblages previously at, say 45°N, while the assemblages once at 55°N would now be found in subtidal muds at 60°N. With a full global network of MPAs designed to cover representative areas of (physical) habitat most assemblages will be able to migrate.

Two critical caveats are (i) that to be effective as a tool protecting 'representative habitats and species assemblage, MPA networks will need to be sufficiently extensive to allow connectivity and hence movement of the assemblages and (ii) the need for biogenic habitats to alter their distribution. The former is an issue explored in the SGMPAN report while there is no obvious reason *per se* why a habitat forming species cannot shift its biogeographic range, there are issues of establishment and timing. The environmental conditions would need to be suitable for establishment of the habitat forming species and development would need to proceed sufficiently to provide habitat for ecologically dependent species before conditions at the former location become untenable.

The other issue highlighted by this thought experiment is what occurs at the ends of the latitudal gradient? In tropical regions such shifting ranges could leave behind an ecologically poor system as species are lost from the pool with no further source of thermally tolerant replacements. While at high latitudes species currently occurring at 89°N might simply go extinct, or at least face very changed ecological conditions through competition with species expanding their range pole-wards. These are issues that can only be solved by addressing the causes of global climate change not through MPA design!

Another issue occurs in enclosed or semi-enclosed seas. In Europe, examples include the Baltic, Mediterranean and Black Seas; in North America - the Gulf of St Lawrence and Hudson Bay. Ecological enclosures may not be formed by land alone; an area of deep water surrounded by shallow water may isolate/enclose deep-water habitats. In these areas, available habitat niches may disappear completely rather than move. The occurrence of relict populations in the present fauna (e.g. the Black Sea harbour porpoise *Phocoena phocoena relictus*) has in many cases been caused by past climate change (glaciations) isolating these populations and habitats. Again, this issue cannot be addressed through MPA network design, but by addressing the causes of global climate change.

On this basis WGECO consider the first practical step to be taken to ensure adequate protection by **representative** MPAs is the establishment of a global network of connected networks of MPAs.

WGECO is aware of the literature that suggest that in many cases species will not only move pole-wards but also in deep water or offshore (into cooler oceanic water masses) (Graham *et al.*, 2008; Greenstein and Pandolfi, 2008; Bongaerts *et al.*, 2010; Tittensor *et al.*, 2010).

Terrestrial conservation bodies are currently discussing the concept of 'assisted migration' where by species are actively transported to and established in new sites (usually within protected areas) when the environmental conditions are expected to remain suitable for them in the medium term (McLachlan *et al.*, 2007). WGECO feels that this approach is unlikely to be feasible in the marine environment except in exceptional cases.

5.2.1.2 Targeted MPAs

MPAs targeted at the protection of particular species or habitat features are a key management tool and much of the consideration of MPAs in the SGMPAN report seems to be implicitly based on a consideration of the challenges of developing MPA networks for such groups. The challenge here is that we can be certain that whatever is established now will not be fit for purpose in a world with a changed climate regime. This applies to species that will, if they can, alter their distributions, moving to deeper water, into oceanic water masses or pole-wards and to habitats where altered temperature, changed storm frequency, altered frontal systems, etc. will alter their suitability for the associated biological assemblages.

The idea of declaring MPAs then reviewing them every few years and relocating them as needed to match the new distribution of the target is simply untenable given the technical and political challenges of MPA designation in most jurisdictions. It creates the caricature image of MPAs being moved across a global map chasing a pod of rare dolphins! SGMPAN were therefore correct to try to give this issue some prominence. WGECO agree that the challenge is to develop a means of predicting where such priority species will move to and to have a designation process in place that allows designation in advance of the need. WGECO is keen to emphasize that for many management objectives MPAs are not the most powerful, or even an effective, tool and other measures such as changes in fishing gears or fishing practices would be more effective and would also be much easier to adapt to changing distributions of the species of concern.

If the area into which the target is expected to migrate is degraded by human activity then there is the further complication of considering the rate and trajectory of recovery (i.e. the resilience of the system) an issue addressed by SGMPAN (see below).

5.2.2 Size spectra changes–foodweb dynamics, ecosystem functioning

Marine foodwebs are often highly complex, with many species each interacting with many others. The defining feature of these webs is that they are strongly size structured, an organism is able to feed on a wide range of prey in a particular size range and in turn is depredated by a range of predators that changes during their lifetime as they grow.

As noted by SGMPAN the size or basal species in warm waters tends to be smaller (McNab, 1971; Brown *et al.*, 1996). So while our thought experiment assumes wholesale replacement of one functioning assemblage with another the reality will be more complex. Some species will, due to their differing physiological requirements, attempt to move before others and as such predators could be faced with establishing in to a system in which their normal prey have not yet migrated and could be faced with prey of an in appropriate size.

There has been a recent growth in interest in the importance of healthy and functioning ecosystems to deliver, what the economists refer to as, ecological goods and services. For many of these functions knowledge of links to underpinning ecological processes is rudimentary. However for some functions there is a clear link to the size structure of the assemblage. Carbon cycling/sequestration links to longevity and this often correlates with body size, larger benthic fauna deliver more bioturbation and hence greater nutrient cycling, food availability for commercial species is influenced by the food value of the available prey, etc.

It is not clear how knowledge of these concerns should influence MPA network design, but clearly a global network of linked MPAs networks will provide the most robust strategy for reducing the risk of catastrophic system failure to deliver goods and services.

5.2.3 Connectivity

WGECO concluded that there were two scales of connectivity that needed to be considered in the design of MPA networks in the face of climate change. There is the issue of linking individual MPAs into local networks then the connecting of these to allow them to respond to the biogeographic range shifts associated with climate change. The former has received considerable attention in the literature going back several decades and drawing on parallels from the terrestrial conservation experience. However it is important recognize major difference in the functioning of marine and terrestrial systems.

Marine systems have much fewer physical barriers to connectivity and given the spatial scale of ocean currents, there is potential for species to be dispersed long-distances. This creates an evolutionary trade-off between the advantages of finding new areas to colonize and being carried away from the favourable conditions that prevail at the parental location. A growing body of evidence indicates that dispersal distances are often relatively short (e.g. Cowen *et al.*, 2000; Cain *et al.*, 2003; Cowen *et al.*, 2006) and several mechanisms are adopted by marine species to limit dispersal. Active behaviour such as vertical migration (Knights *et al.*, 2006; Wilson White *et al.*, 2010), in response to chemical cues or attractants from adult conspecifics (Jeffery, 2000; Pawlik, 1992) allows a dispersing juvenile to utilize local discontinuities or spatially differentiated flows to limit or facilitate transport. Such behavioural mechanisms have the potential to strongly influence population and community dynamics, and evolution by affecting rates of supply, and timing and density of recruitment (McQuaid and Phillips, 2000, Kritzer and Sale, 2004). The density of juveniles can determine the strength of interactions between the juvenile and its environment. Intra- and interspecific competition, predation and resources (i.e. space and food) (e.g. Connell, 1961; Paine, 1966; Firth *et al.*, 2009; Knights and Walters, 2010) are all key processes which, either solely or in combination, can determine rates of population growth and extinction (Brown and Kodic-Brown, 1977; Hanski and Ovaskainen, 2000; Hanski and Simberloff, 1997) and modify ecosystem structure and functioning (Connell, 1961; Emmerson *et al.*, 2004).

Dispersal is a primary driver of species distribution patterns (Nathan and Muller-Landau, 2000; Levin *et al.*, 2003). However, dispersal distance and the factors affecting recruitment are variable over spatial and temporal scales and the outcome of the complex interactions between a species and its environment which determine likelihood of population establishment and persistence are poorly understood for the majority of species. Thus, the distance between MPA areas that would be required to support an ecologically coherent network is species dependent and the use of potential dispersal distance alone does not necessarily imply a coherent network will be achieved. Furthermore, availability of habitat does not necessarily ensure successful establishment as other factors such as resource availability (i.e. space and food) may result in high rates of mortality (Jenkins *et al*., 2008). Therefore, characterization of individual species life-histories and evaluation of the complex interactions that affect recruitment success must be considered when developing coherent MPA networks so they are appropriate to its objectives.

As population sizes decrease in the face of human pressures, fishing, pollution, and climate change, the likelihood that sufficient propagules will arrive to a habitat in sufficient numbers to dominate interactions so that they become established may reduce (Underwood and Keough, 2001; Knights and Walters, 2010). Some species capable of dispersing rapidly over distances sufficient to escape those pressures may be capable of quickly responding to changes in environmental conditions i.e. Mobile Link Organisms (*sensu* SGMPAN, 2011) and an MPA network may not be overly beneficial. However, for species where dispersal distance and recruitment success are relatively low, the development of a coherent network may facilitate persistence.

5.2.4 Resilience

The SGMPAN report provides a detailed consideration of the issues around the resilience of marine 'ecosystems' in the face of climate change and summarizes much of the relevant literature. However, it is unclear from the report how this is seen as influencing MPA network design. WGECO reflected on the science and the challenge and concluded that the issue here relates to the need to understand the recovery of sites that are in degraded states to a more favourable condition that could meet the objectives of an MPA.

For example if one has a species of conservation concern and an MPA is declared around a breeding area, what happens when conditions warm and the species abandons this site? How quickly will a degraded site nearby recover to a state to allow breeding on it? WGECO was unconvinced on the practical applicability of this implied approach. How can we predict where a species might move to? How can it be given MPA status when it does not currently support the species? How can we choose sites to recover?

There are many national and international programmed currently targeted on promoting the recovery of degraded populations and habitats (e.g. Canadian Oceans Act, EU MSFD, see Section 6 (ToR D)).

5.3 Climate change scenarios

WGECO considered the SGMPAN report would benefit from inclusion of material relevant, but specific, to the NE Atlantic region. This section considers the implications of various climate change scenarios on the marine environment of the NE Atlantic.

How species respond to climate change is largely unknown. Changes in distribution, phenology and abundance have been described (Hughes *et al*., 2000; Parmesan and Yohe, 2003) and can have positive or negative consequences for the persistence of a species. Not surprisingly, commercially and economically important species and those species that are significantly below their reproductive potential have been of primary focus when attempting to understand responses to climate change.

A dramatic increase in global sea temperatures in the mid-20th century has been closely correlated with anthropogenic greenhouse gas concentrations (IPCC, 2007) and the effect of global increases in temperature is likely to have wide-ranging consequences for marine species (Ottersen *et al.*, 2001). Changes in temperature have the potential to affect large-scale oceanic processes including advection and convection patterns, evaporation and precipitation (Tasker, 2008) and the complex interaction between the ocean and atmosphere can influence the ecological processes they affect. It is also clear that populations suffering from the adverse effects of human activities are more vulnerable to the effects of climate variation (Hsieh *et al.*, 2006).

5.3.1 Climate predictions

There is considerable uncertainty in climate model predictions and disagreement between emission scenarios (Figure 4.3.1.1) and the IPCC Fourth Assessment Report (IPCC, 2007) describes over 40 special report emissions scenarios (SRES). The projections available from IPCC (2007) are probabilistic and highly smoothed through the compositing of ensembles of simulations from multiple Atmosphere Ocean General Circulation Models (AOGCMs). While clear and apparently robust spatial and temporal patterns are apparent for many variables, they are generally on large space scales with relatively monotonic temporal changes because of the compositing and spatial smoothing, coupled with a poor understanding of natural variability (ICES, 2011).

To address these limitations, other techniques are available to provide climate change projections including dynamical and statistical spatial “downscaling” techniques (e.g. Hayhoe *et al.*, 2008). However, these techniques are less sensitive to small magnitude changes and therefore, generally most useful in detecting large magnitude anthropogenic changes over multiple decadal time-scales and improving our understanding of long-term natural variability.

5.3.1.1 Global oscillations

Changes in climate are predicted to greatly impact the functioning of our oceans and regional seas. Changes in temperature may affect large-scale ocean processes e.g. the El Niño-Southern oscillation (ENSO) and the North Atlantic oscillation (NAO) and global phenomena such as melting ice caps and rising sea temperatures may induce a change in the state or behaviour of those oscillations (Lu and Greatbatch, 2002) which in turn may greatly impact on marine habitats and species.

Global climate models that are used to study anthropogenic climate change do not yet give unequivocal predictions for the future of ocean oscillations. For example, it is uncertain how the NAO will respond to climate change, as there is a strong link between the NAO and Atlantic storm tracks, and it remains unclear how storm frequency and intensity might change. Nevertheless, worst-case scenario predictions suggest a reduction in NAO flow. Fluctuations in large-scale climate phenomena may produce large effects at various trophic levels (Post *et al.*, 1999). Even small changes in the state of ocean oscillations are expected to result in disproportionately large impacts extending beyond the extent of the oscillation itself (Allan *et al.*, 1996) and greatly impacting marine and terrestrial ecosystems (Ottersen *et al.*, 2002).

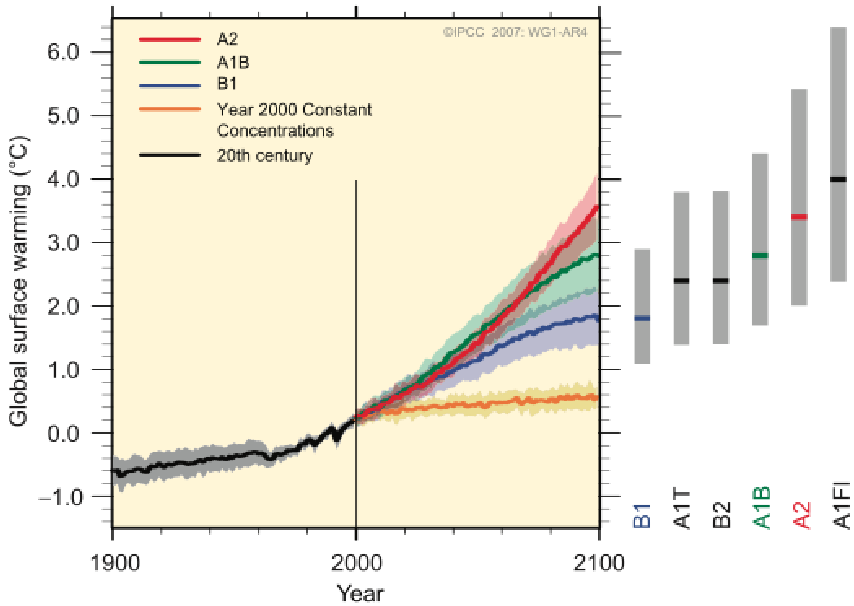

Figure 4.3.1.1. Solid lines are multi-model global averages of surface warming (relative to 1980–1999) for the scenarios A2, A1B and B1, shown as continuations of the 20th century simulations. Shading denotes the ±1 standard deviation range of individual model annual averages. The orange line is for the scenario where concentrations were held constant at year 2000 values. The grey bars at right indicate the best estimate (solid line within each bar) and the likely range assessed for the six SRES marker scenarios. The assessment of the best estimate and likely ranges in the grey bars includes the AOGCMs in the left part of the figure, as well as results from a hierarchy of independent models and observational constraints. Redrawn from IPCC AR4 (2007).

5.3.2 Linking the NAO and European climate

The NAO is generally considered to be part of larger-scale patterns of climate variability at mid to high latitudes in the northern hemisphere such as the Arctic Oscillation (AO) and the Northern Annular Mode (NAM). Described as a change in pressure between the subtropic atmospheric high-pressure zone centred over the Azores and the atmospheric low-pressure zone over Iceland, the NAO is the dominant mode of atmospheric behaviour in the North Atlantic (Hurrell, 1995) and is seen to influence the distribution and fluxes of major water masses and currents in the Atlantic (Dickson, 1997; Curry *et al.*, 1998; Reid *et al.*, 1998).

The NAO influences winter climate in the North Atlantic, accounting for approximately 50% of the increase in winter temperature throughout the extra-tropical northern hemisphere (Hurrell and van Loon, 1997) and can be attributed to nearly all of the cooling in the NW Atlantic and the warming across Europe since the 1970s. The NAO is primarily a winter phenomenon and correlations between the NAO index and winter sea surface temperature (SST) and/or wind strength are apparent (Ottersen *et al.*, 2001; Borges *et al.*, 2003).

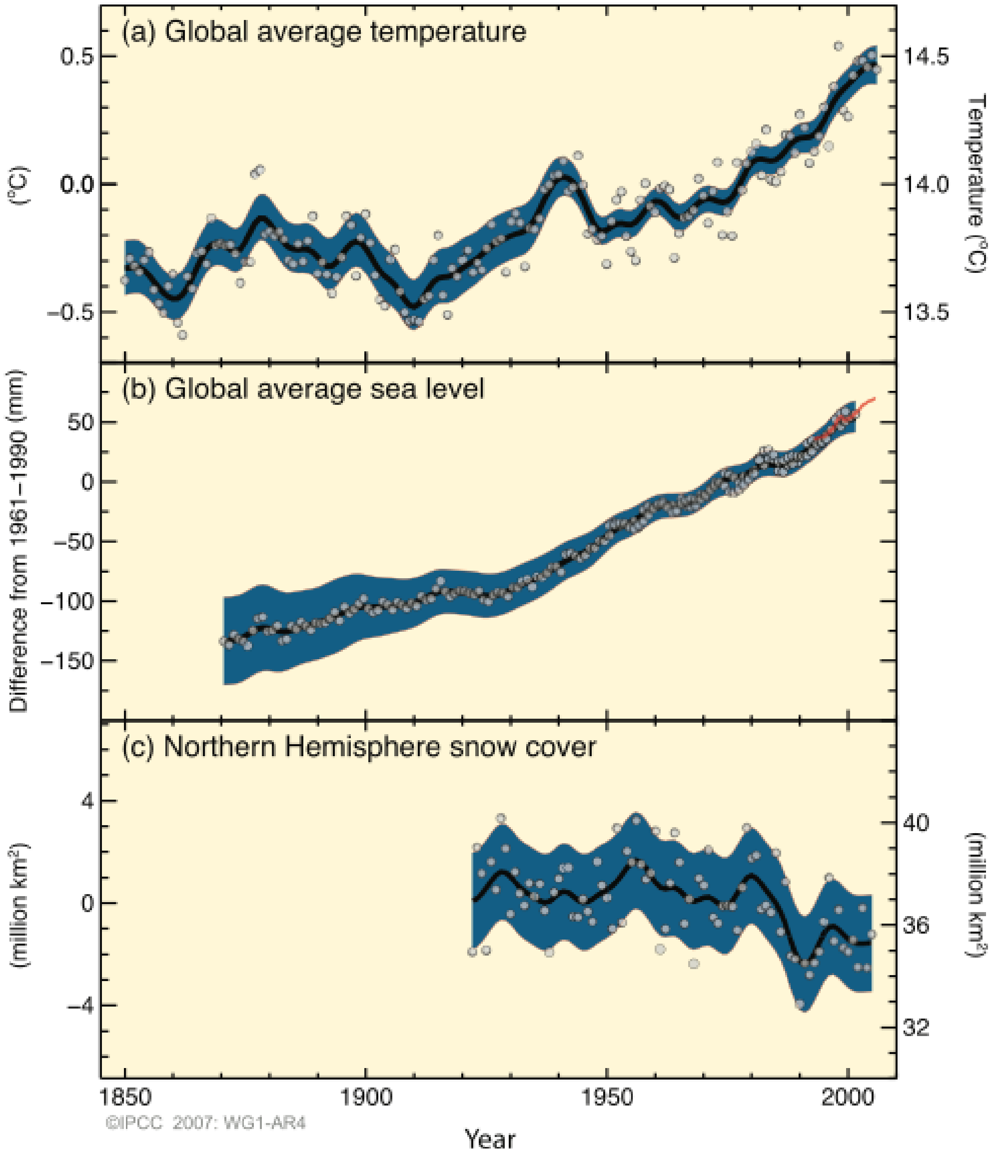

Figure 4.3.1.1.2. Observed changes in (a) global average surface temperature, (b) global average sea level from tide gauge (blue) and satellite (red) data and (c) northern hemisphere snow cover for March–April. All changes are relative to corresponding averages for the period 1961–1990. Smoothed curves represent decadal average values while circles show yearly values. The shaded areas are the uncertainty intervals estimated from a comprehensive analysis of known uncertainties (a and b) and from the time-series (c). Redrawn from IPCC AR4 (2007).

5.3.3 Sea surface temperature and seabed temperature trends

The North Atlantic Oscillation and the Gulf Stream indices have increased, peaking in 1995 with strong negative values in 1985 and 1996 (Dulvy *et al.*, 2008). In conjunction, sea surface temperatures (SST) have continued to increase globally (BSH 2010; NOAA 2010) although large annual fluctuations are apparent in any given year (Figure 4.3.3.1). Within the NE Atlantic and surrounding seas, two 'points of change' in the rate of temperate increases year-on-year are apparent. Firstly, between 1920 and 1935, average SST rose by approximately 0.2°C (~0.01°C per annum) from the previous average of 10.9°C to 11.1°C, followed by a 50-yr period of relative stability (Figure 4.3.3.1). Secondly, since the early 1980s, the rate at which SST increased year-on-year accelerated to ~0.02°C per annum corresponding to an increase in SST of ~0.8°C over the 30-yr period between 1980 and today (Figure 4.3.3.1). This rate change has been mirrored in annual mean SST observations for the Atlantic Ocean (NOAA, 2010).

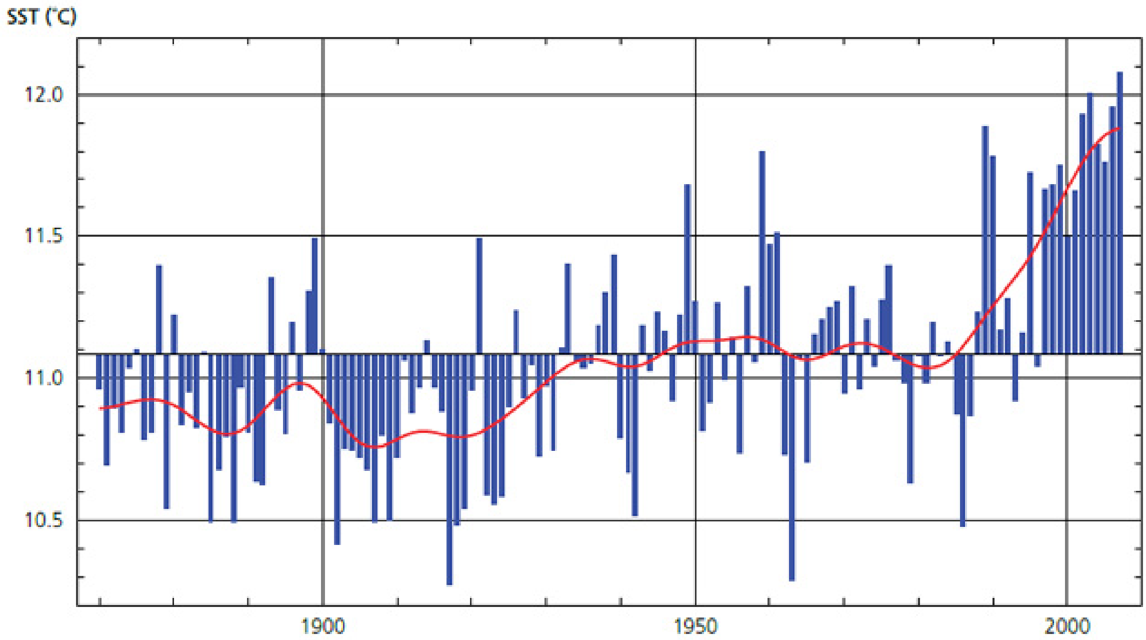

Figure 4.3.3.1. Annual average sea surface temperature (SST) for the Atlantic Ocean (top) and UK territorial waters (bottom; HadISST dataset), between 1984 and 2010 and 1870 and 2007 respectively. Blue bars show deviates of the annual average from the 1961–1990 average, and the red line shows annual averages after smoothing with a 21-point binomial filter (after Rayner *et al.*, 2003). (Source: NOAA, 2010; Charting Progress 2, DEFRA).

North Sea winter bottom temperatures have also risen, on average by 1.6°C over 25 years, with a 1°C increase occurring between 1988–1989 alone (Figure 4.3.3.2). However, this is interspersed with some periods of cooling in localized areas (e.g. Neumann *et al.*, 2009; Dmitrenko *et al.*, 2009). The warming bottom temperatures have coincided with a long-term shift towards a positive NAO phase, a northward shift in the Gulf Stream and stronger Atlantic inflow into the northern North Sea (Dulvy *et al.*, 2008).

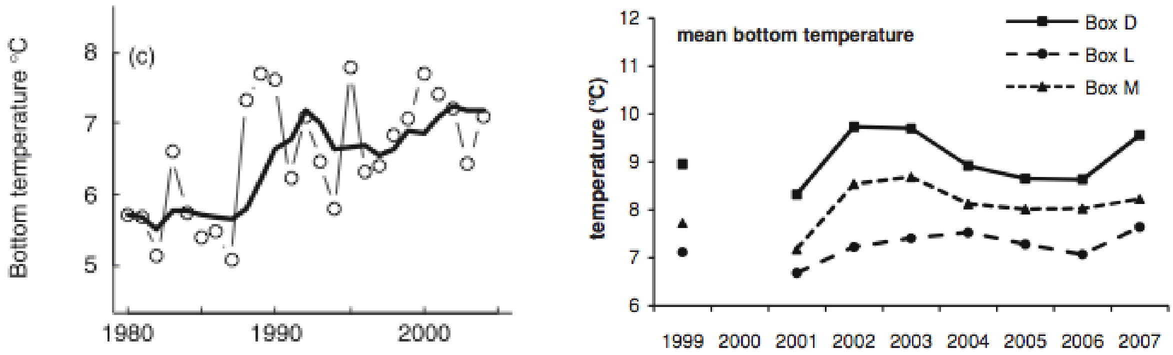

Figure 4.3.3.2. Mean bottom temperature from (a) the North Sea and Northeast Atlantic (1980–2004) (Dulvy *et al.*, 2008), and (b) localized periods of cooling in the northern North Sea (Neumann *et al.*, 2009). Annual values in (a) are represented by the connected points with the 5-year right- aligned running mean represented by the bold line.

5.4 Climate change impacts on marine species

WGECO considered the SGMPAN report would benefit from inclusion of material relevant, but specific, to the NE Atlantic region. This section considers the important ecosystem components of the marine ecosystem in the NE Atlantic.

The importance of temperature in regulating the behaviour and dynamics of marine species and its relationship with oceanographics is well documented. Temperature acts on almost every biological step of a species' life history including growth (Brander, 1995), maturity (Tyler, 1995), reproduction (Hutchings and Myers, 1994; Kjesbu, 1994), food availability (Reid *et al.*, 1998; Loeng *et al.*, 1995; Ottersen and Loeng, 2000) and larval growth and mortality (Pepin, 1990; Otterlei *et al.*, 1999). For example, an increase in temperature is expected to move the timing of the spawning migration by anadromous species into freshwater forward (e.g. Quinn and Adams, 1996) also resulting in the earlier out migration and changes in survival of juveniles (Boisneau *et al.*, 2008; Perry *et al.*, 2005; Southward *et al.*, 2004). Many extant North America and northern European species experienced changes in distribution during the last glacial period. Examination of the drivers for the advance and retreat of these taxa in the face of global warming and cooling could provide insights into their response under current and future climate scenarios.

These changes can affect the functioning of foodwebs by modifying core components such as rates of primary production and modifying species composition. For example, alterations of oceanic circulation patterns associated with interannual temperature variation can result in unfavourable conditions for the copepod *Calanus finmarchicus*, and in some cases, have led to a significant decrease in the abundance of the species. In those years when conditions have been unfavourable to *C. finmarchicus*, hydroclimatic shifts have proven beneficial to other species of copepod e.g. *C. helgolandicus* abundance increases during the 1980–1990s (Fromentin and Planque, 1996; Planque and Fromentin, 1996).

The general link between spatial and temporal scales in nature makes it difficult to reveal the effects of short-term (e.g. interannual variability) climate changes on species distributions simply because such changes are not correlated across larger spatial scales. Analysis of changes, even within an explicit spatial context such as the Northeast Atlantic, requires observations at decadal time-scales and longer.

5.4.1 Effects of climate change on key species and habitats in the Northeast Atlantic

Within the ICES framework, a consortium of scientists reviewed the evidence of the effects of climate change (e.g. long-term datasets) on the distribution and abundance of marine species in the OSPAR Commission Maritime Area (Tasker, 2008). This work focused primarily on changes with respect to changes in sea surface temperature. Table 4.4.1.1 shows a list of the key species identified for zooplankton, fish, seabirds and marine mammals in the whole OSPAR region that are known to respond to climate change, or are likely to do so (Tasker, 2008). The table does not contain those species and habitats that are already mentioned in the SGMPAN 2011 report. WGECO considers that the evaluation with respect to zooplankton, fish, seabirds and marine mammals is very thorough and for that reason does not consider more information is needed. This table does not include benthic organisms as WGECO considered these require more extensive evaluation (see Section 4.2.2).

Table 4.4.1.1. Key species of zooplankton, fish, seabirds and marine mammals in the OSPAR region (Tasker, 2008) absent from the Northwest Atlantic region.

Scientific name	Distribution
ZOOPLANKTON	
Centropages typicus	Northwest and Northeast Atlantic
Candacia armata	Species can be found in oceanic and neritic water, but their abundance is higher alongshelf edges generally until about 55ºN
Calanus helgolandicus	Species can be found in oceanic and neritic water, but their abundance is higher alongshelf edges generally until about 55ºN
Pseudocalanus acuspes	European Waters
Mnemiopsis leidyi	Northwest and Northeast Atlantic
Temora stylifera	Northwest and Northeast Atlantic. African waters
Aurelia aurita	Northwest and Northeast Atlantic
Cyanea lamarcki	Northeast Atlantic
Cyanea capillata	Northwest and Northeast Atlantic
Aglanta digitale	Northwest and Northeast Atlantic, Artic, Pacific, Mediterranean Sea
FISH	
Alosa fallax	Northeast Atlantic, Mediterranean Sea and Black Sea
Anarhichas lupus	Northeast Atlantic, Northwest Atlantic and Mediterranean
Arnoglossus laterna	Eastern Atlantic: Norway to Angola. Also known from the Mediterranean and Black Sea
Belone belone	Eastern Atlantic and Mediterranean Sea
Buglossidium luteum	Northeast Atlantic and Mediterranean
Callionymus spp.(lyra)	Northeast Atlantic and Mediterranean
Capros aper	Eastern Atlantic and Mediterranean
Engraulis encrasicolus	Eastern Atlantic, Mediterranean, Black and Azov seas
Entelurus aequoreus	Eastern Atlantic: Iceland and Norway to Azores and also enters Baltic Sea.
Eutrigla gurnardus	Eastern Atlantic: Norway to Morocco, Madeira, and Iceland. Also known from the Mediterranean and Black Sea (Ref. 4697).
Galeoides decadactylus	Eastern Atlantic
Glyptocephalus cynoglossus	Northwest and Northeast Atlantic

Scientific name	Distribution
Hippoglossoides platessoides	Northwest and Northeast Atlantic
Lepidorhombus whiffiagonis	Northeast Atlantic: Iceland southward to Cape Bojador (26°N), West Sahara and in the western Mediterranean.
Limanda limanda	Northeast Atlantic
Lophius piscatorius	Eastern Atlantic
Melanogrammus aeglefinus	Northwest and Northeast Atlantic
Merlangius merlangus	Northeast Atlantic
Merluccius merluccius	Eastern Atlantic. Also in the Mediterranean and the Balck Sea
Micromesistius potassou	Northwest and Northeast Atlantic
Microstomus kitt	Northeast Atlantic
Molva molva	Northwest and Northeast Atlantic. Also in the Mediterranean Sea
Mullus surmulletus	Eastern Atlantic, Mediterranean and the Black Sea.
Petromyson marinus	Northwest and Northeast Atlantic.
Phycis blennoides	Eastern Atlantic and Mediterranean Sea
Pleuronectes plateas	Northeast Atlantic and Mediterranean Sea
Pollachius virens	Northwest and Northeast Atlantic.
Raja clavata	Eastern Atlantic, Mediterranean and the Black Sea
Leucoraja naevus	Eastern Atlantic and Mediterranean Sea
Rhinonemus cimbrius	Northwest and Northeast Atlantic.
Sardina pilchardus	Northeast Atlantic, Mediterranean and the Black Sea.
Scomber scombrus	North Atlantic.
Scyliorhinus canicula	Northeast Atlantic
Solea vulgaris / Solea solea	Eastern Atlantic and Mediterranean Sea
Sprattus sprattus	Northeast Atlantic: North Sea and Baltic south to Morocco; also the Mediterranean, Adriatic and Black seas.
Squalus acanthias	Northwest and Northeast Atlantic, Mediterranean and Black Sea
Trachurus trachurus	Eastern Atlantic and Mediterranean Sea
Trisopterus esmarki	Northeast Atlantic
Trisopterus luscus	Eastern Atlantic and Mediterranean Sea
Trisopterus minutus	Eastern Atlantic andMediterranean Sea
Zeus faber	Worldwide in distribution
MARINE MAMMALS	
Tursiops truncates	Warm and temperate tropical oceans worldwide
Delphinus delphis	Warm-temperate portions of the Atlantic and Pacific Oceans.It is also found in the Caribbean and Mediterranean Seas.
Lagenorhychus albirostris	North Atlantic Ocean
Phocoena phocoena	North Atlantic, North Pacific and the Black Sea
SEABIRDS	
Rissa tridactyla	North Pacific and North Atlantic oceans
Sterna paradisaea	Arctic and Subarctic regions of Europe, Asia, and North America
Puffinus mauretanicus	Mediterranean and adjacent Atlantic
Uria lomvia	Northernmost areas of the North Atlantic and Pacific
Phalacrocorax aristotelis	Western and southern Europe, southwest Asia and north Africa

5.4.2 Benthic communities

The Section 6.2 of the SGMPAN report identifies those benthic species and habitats that are of conservation concern or are considered to play an important role in the ecosystem in the NW Atlantic (US and Mexico). These species include various reef- and non-reef forming cold-water scleractinian corals, sponges and molluscs (both habitat forming and targeted). The report mentions that increasing temperatures can intensify ocean acidification, thus affecting species whose structural support is mainly calcium carbonate (e.g. cold-water corals and sponges) and other species that may become more susceptible to diseases. Examination of the potential effects of climate variation on benthic organisms and consideration of the various managing options in response to such climate effects received less attention. The effects of climate change on the distribution and abundance of the key zooplankton, benthos, fish, seabird and marine mammal species were evaluated for the OSPAR maritime area (Tasker, 2008).

The effects of climate variation on benthic communities in the NE Atlantic are probably best understood in the intertidal zone and least understood in the deep sea, hence reflecting the practicality in studying these habitats. Hawkins *et al*. (2009) and Wethey *et al*. (2011) reviewed the various changes that have occurred in the patterns in spatial distributions and abundance of intertidal organisms in the waters around UK, Bay of Biscay and off the Iberian Peninsula, based on long (>50 years) time-series. These studies show a number of well documented cases of changes in abundance and distribution in response to increasing temperature. This includes replacement of the barnacle *Semibalanus balanoides* by *Chthamalus* species, which is gradually moving to higher latitudes. For the subtidal, much less data on long-term trends of benthic invertebrates are available. Some studies have shown that growth rates of bivalves (based on age measurements), such as for *Arctica islandica* (Wanamker *et al*., 2009), *Clinocardium ciliatum* (Carroll *et al*., 2011) and *Serripes groenlandicus* (Ambrose *et al*., 2006) were related to temperature. Bivalves may therefore be considered as ideal indicators of climate change on multidecadal scales (Ambrose *et al*., 2006). There have been few studies where samples were collected on multidecadal scales. Frid *et al*., (2009) analysed data collected off the UK that spanned 36 years. They concluded that the observed changes in the benthic community structure could be driven by a combination of fishing and climate variation.

Changes in the spatial distributions of fisheries targeted invertebrates in response to the variation in climate may have various implications for their management. Shephard *et al*. (2010) related increases in *Pecten maximus* density around Isle of Man with increasing temperature and this was related to increased food availability, which in turn resulted in greater gamete production. A converse pattern was observed for the scallop *Chlamys islandica* in Icelandic waters where stocks have collapsed or their distribution limits have been altered (Guijarro-Garcia, 2006). Several causes for the decline are likely including increased seawater temperature, intense fishing pressure, diseases and potentially altered food availability. Herraiz *et al*. (2009) suggested that catches of *Nephrops norvegicus* could be related to the NAO index. Furthermore, an increase in temperature can alter the interactions among target species. Biomass of cod has been suggested to be inversely related to *Pandalus borealis*, which is an important prey for cod. Parsons (2005) suggested that the decrease in cod during cold winters resulted in greater abundance of *P. borealis*.

Ocean acidification as a consequence of ocean warming is a serious concern for deep-sea organisms that form calcareous shells or skeletons (ICES, 2010). During the calcification process, the two natural polymorphs of calcium carbonate ($CaCO_3$), calcite and aragonite are secreted in the form of shells or skeletons by many organisms, such as cold-water corals and sponges. The level of saturation of calcite and aragonite decreases with increasing depth and in areas with fast CO_2 uptake, such as NE Atlantic (Turley *et al*., 2010). The depth level at which waters are undersaturated by aragonite and calcite is gradually moving to shallower waters and predictions suggest this trend is to continue (Turley *et al*., 2010). Some habitat forming species, such as cold-water corals found in undersaturated water can therefore be at considerable risk. There are data suggesting that climate driven changes at surface waters can affect carbon cycling in the deep sea (Turley *et al*., 2010; Wood *et al*., 2011).

Predicting responses of a given species to climate variation would always be a challenging task. In some cases, there are sufficient data to forecast trends with a reasonable degree of certainty. Hawkins *et al*. (2009) forecasted trends of the barnacle species *Semibalanus balanoides* and *Chthamalus* with high certainty. Gröger and Rumohr (2006) modelled and forecasted the long-term trends of the macrobenthic community in the Western Baltic. They concluded that the trends in species richness were strongly influenced by the winter NAO. In many cases, data on long-term trends are not available and other approaches are required, such as evaluation on species traits (e.g. life history), could provide insights into how those species would respond to changes in climate. Considering the high species richness in most benthic communities, a more manageable approach is to select a subset of species that are considered to play a key role; an approach that could be considered by SGMPAN. Such an exercise was undertaken in the European EFEP project, where species and habitats in the North Sea were evaluated based on criteria of economic, societal, ecological and functional importance (Ragnarsson *et al*., 2004) (Table 4.4.2.1). The economic importance was evaluated based on direct monetary value of the target species (e.g. *Nephrops norvegicus*). Species and habitats of conservation concern were considered important under the societal importance criteria (e.g. *Lophelia pertusa*). Species of functional importance were principally those that were important in modifying the biogeochemical environment (e.g. bioturbators) and/or the physical and biological environment (e.g. habitat formers). Ecological importance of a species was evaluated based on its importance in trophic relationships, either as a prey or predator. The evaluation was carried out based on an extensive literature review. From the original list of approximately 1500 species (Künitzer *et al*., 1992; Callaway *et al*., 2002), 57 species could be evaluated, of which, 27 identified as playing a key role in the North Sea ecosystem. The majority of these species have extensive biogeographic distributions but local and/or regional populations may be fragmented. As an example, the stony coral *Lophelia pertusa*, is ubiquitous worldwide but is confined to locations with specific environmental characteristics (e.g. geological settings, current strength, substratum type). It is therefore considered of local importance. Loss of these species in response to environmental changes e.g. increase in seawater temperature, could have various consequences and modelling could be used to explore these effects further. Example scenarios for exploration include: '*How does an increase in seawater temperature affect food availability for suspension-feeders?*' and '*How would the loss of an important prey species affect its predators*?'

Table 4.4.2.1. Benthos and habitats selected for the significant web in the North Sea in the European Fisheries Ecosystem Plan (EFEP) project (Ragnarsson *et al*., 2004) Information on the geographical distribution of species based on (http://www.marine species.org and http://www.marlin.ac.uk) in addition to other sources that are cited.

Key invertebrate species	Approximate geographical distribution boundaries
Annelida	
Chaetopterus variopedatus	NW Atlantic and NE Atlantic
Sabellaria spp.	Wordwide
Serpula vermicularis	Widely distributed (except Arctic and Antarctic), (Hartmann-Schröder, 1996); Shetlands, west and south coasts of Britain, western Ireland, and Channel Isles; often abundant (Nelson-Smith *et al*., 1990).
Lagis koreni	East Atlantic from the Barents Sea to Namibia, Mediterranean, Adriatic and Black Seas, Channel, North Sea, western Baltic Sea (Hartmann-Schröder, 1996)
Lanice conchilega	Northern hemisphere, the Channel, North Sea, Danish Straits (Hartmann-Schröder, 1996); around all coasts of Britain and Ireland but also found off Africa.
Arthropoda	
Callianassa subterranea	South coasts of British Isles, common; elsewhere southwards, in the Mediterranaean (Moyse and Smaldon, 1990), abundant in the southern North Sea (Witbaard and Duineveld, 1989; Rowden and Jones, 1994).
Crangon crangon	Mainly NE Atlantic such as from British isles to SW Africa and Mediterranean.
Jaxea nocturna	From England south to the Mediterranean, particularly in the Adriatic (Moyse and Smaldon, 1990).
Nephrops norvegicus,	From Norway and Iceland to Morocco and Mediterranean; all coasts of the British Isles, common (Moyse and Smaldon, 1990).
Pandalus borealis	NE and NW Atlantic. In NE Atlantic ranges from Svalbard to NE England
Carcinus maenas	Worldwide distribution
Upogebia deltaura	Southern Norway to Spain and Mediterranean, and Black Sea, perhaps all coasts of British Isles, common (Moyse and Smaldon, 1990).
Cnidaria	
Lophelia pertusa	Widely found in NE Atlantic but also in NW Atlantic, Pacific and Indian ocean
Pennatula phosphorea	NE Atlantic (Iceland to Portugal, Mediterranean,) New Zealand, NW Atlantic?
Echinodermata	
Amphiura filiformis	Norway to the Mediterranean; common off all British coasts.
Asterias rubens	Abundant throughout the NE Atlantic, from Arctic Norway, along Atlantic coasts and off Africa whereas only found occasionally in the Mediterranean (Mortensen, 1927); abundant on all British coasts (Moyse and Tyler, 1990). Found also widely in the NW Atlantic.
Ophiotrix fragilis	Found widely in the NE Atlantic, from Norway and Iceland to French waters, also off S Africa
Ophiura spp.	NW and NE Atlantic
Brissopsis lyrifera	Distributed from Norway and Iceland to South Africa and the Mediterranean, also present on the east coast of North America but not Greenland; recorded off the west, north and east coasts of the British Isles, but not off the south coast. Found also in the NW Atlantic.

Key invertebrate species	Approximate geographical distribution boundaries
Echiura	
Maxmuelleria lankesteri	NE Atlantic subregions e.g. Irish Sea, west Scotland, Kattegat, Skagerrak (Knight-Jones and Ryland, 1990, Hughes *et al*., 1996).
Mollusca	
Buccinum undatum	Distributed from Iceland and northern Norway to Portugal; common and often abundant around British Isles, except Scilly Isles (Hayward *et al*., 1990). Sometimes present in brackish waters (Ager, 2003b). Found in widely off NW Atlantic, such as US and Canada.
Cerastoderma edule	East Atlantic from northern Norway and south to Morocco and recorded in Africa as well.
Mytilus edulis	In the East Atlantic, it ranges from Arctic waters south to the Mediterranean; widespread and common on all British coasts (Hayward, 1990). Found widely throughout the NW Atlantic including US and Canada.
Modiolus modiolus	Similar distribution as for Mytilus edulis but does not reach so far south.
Ostrea edulis	NW and NE Atlantic and Mediterranean. NE Atlantic from Norway south to the Mediterranean (Hayward, 1990)
Pecten maximus	East Atlantic from the British Isles to Morocco
Macoma balthica	NE Atlantic from UK to Portugal and Baltic and in NW Atlantic

5.5 Conclusions

Having reviewed and reflected on the work of SGMPAN WGECO have concluded that there are two areas where further developments are needed. WGECO would be keen to work with SGMPAN on moving this forward.

International agreements (CBD) are pressing for designation of networks of MPAs to protect representative areas of habitat. WGECO have shown that local networks of MPAs should ideally be nested within a larger global network connected by dispersal pathways. Such a network should contribute to enabling a change in species and habitat distribution to proceed and also allow access to refugia in deeper water or offshore water masses.

However, fishing and other anthropogenic activities may precondition a system in such a way that the rate of movement and the establishment of new colonizers may be altered. MPAs may relieve pressures on new colonizers. A network that is resilient to the effects of climate change may be achieved through protecting representative habitats along the expected change gradient. Physical oceanographic climate-changes in substrata (e.g. mediated through storm events in coastal areas) or circulation (e.g. movement of the Gulf Stream) would influence direction of colonization, while other factors such as temperature and salinity changes would influence the timing of colonization events. To a large extent a sufficiently extensive and global network will alleviate many of these concerns.

It is now widely recognized that marine invertebrates often have short realized dispersal distances, for example compared to pelagic fish larvae. Therefore, precautionary design criteria should assume connectivity through larval dispersal over appropriately short distances. While the local networks provide for redundancy and protection against unpredictable events (oil spills, disease out breaks) there should also be strong connections. While we might envisage a gradual movement of taxa in the face of changing environmental conditions, hydrographic features and current biogeographic boundaries might act as barriers to this movement until they undergo sudden catastrophic change. **WGECO therefore recommends that MPAs designed to include representative areas should be particularly concentrated around range boundaries set by hydrographic features which may be affected by climate change such as frontal systems.**

WGECO recognize that much of the SGMPAN report focused on the issues associated with predicting changes in distribution of mobile species. **WGECO would encourage steps to build a greater understanding of the 'behaviour' of species of particular conservation concern in the face of climate change.**

WGECO was tasked with reviewing the SGMPAN report (2010). In doing so, it was felt that WGECO could contribute to the discussions dealing with protection of key species and habitats on a shifting biogeographic baseline. For this to be useful to SGMPAN, a related ToR should be considered during the WGECO 2012 meeting, which precedes the final meeting of SGMPAN. With this in mind, WGECO would be happy to receive a ToR from SGMPAN for its 2012 meeting.

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6 ToR D: Marine Strategy Framework Directive

This ToR is in two parts;

- Identify elements of the WGECO work that may help determine status for the 11 Descriptors set out in the Commission.
- Provide views on what good environmental status (GES) might be for those descriptors, including methods that could be used to determine status.

Several previous reports of WGECO have provided material and discussion that support the indicator selection and target setting processes that are required for the MSFD. Many of the points which refer to the selection of thresholds and the relationship between pressures and target achievement will be reviewed and developed further in ToR e) Reviewing the outputs of the ICES Strategic Initiative on Biodiversity Advice and Science (SIBAS) Workshop on "Biodiversity indicators for assessment and management". To avoid overlap with ToR e), this section will deal only with those elements of previous work that directly support status assessment, specifically definitions of thresholds, targets and the terminology used in different legislation.

The second part of the ToR will be addressed using a summary of current European Member State approaches to GES assessment, and a review to show existing best practice, suggestions for best tactical use of existing knowledge, and links to further work through the more strategic development under ToR e). Particular effort has been invested in a review of options for extended, comprehensive and integrated monitoring programmes following an assessment undertaken by ICES Workshop on Cataloguing Data requirements from surveys for the Ecosystem Approach to Fisheries Management (WKCATDAT) to review novel datasets that might be collected during routine surveys and contribute to MSFD indicators.

While reviewing current best practice by Member States, WGECO have identified that there are significant hurdles still remaining to allow the translation of concepts of sustainable use into practical indicators and thresholds. This includes a comprehensive and structured approach to GES assessment. Issues highlighted in the comparison between national approaches highlight how interpretation of GES by one Member State can be incompatible with that of another neighbouring nation. For example, work to integrate a suite of biodiversity state indicators into a single measure that responds to management action is laudable, however the process for ensuring that this is consistent with other approaches in the same subregion is unclear, and Regional Seas Conventions have a role to play.

6.1 Introduction

The completion of a marine strategy is a priority for EU Member States (MS), who are required to provide indicators of GES to the European Commission by July 2012, using the guidance provided in the Commission Decision document (2010/477/EU). Parallel activities related to monitoring programmes, economic assessment and necessary management measures have been included to varying extents by Member States to support the development of GES indicators and targets. These activities describe a range of approaches from MS that are developing biodiversity status indicators and targets, including those using a theoretical basis using ecological criteria, and those which rely on sustainable levels of exploitation to infer acceptable levels of impact. Consideration of datasets that are already available through existing surveys, or those which can be readily added, can also affect judgements on indicator selection.

WGECO has commented in previous reports on generic methods for target setting which will be appropriate to all of the descriptors of GES. These observations, together with additional points of relevance are covered further in Section 6.2 below.

Of the 11 Descriptors of GES, WGECO has experience and expertise in those biodiversity indicators specifically linked to fishing effects. WGECO provides generic comments relevant to all descriptors based on previous deliberations, but for detailed description has limited its comments to the following Descriptors;

> 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.
>
> 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.
>
> 4) All elements of the marine foodwebs, 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.
>
> 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.

EU Member States are already making progress with the selection of indicators and targets for these GES descriptors. During review of this progress several themes came to light that were common to all activities but which were being approached with different emphasis.

- The first was the extent to which current monitoring activity and routinely available national data were influencing the selection of indicators and the confidence with which targets were being set. In some cases it was apparent that lack of data, and the high cost of initiating new dataseries, was limiting the enthusiasm for choosing new indicators.
- Two following areas of common interest was the selection of indicators themselves and the consistency with which MS were interpreting them, and the extent to which common targets and reference points were being selected.
- Finally, it is evident that MS are taking different account of the management measures necessary to achieve the targets. Such assessment is necessary for the cost benefit analysis that will be required in the Initial Assessment. Economic consequences can either be evaluated once targets have been set, or can be used as an integral part of the target setting process, influencing the outcome. Under each of these scenarios the final targets selected may be different.

The following sections describe progress under these three categories by the United Kingdom and by Germany. Other approaches suggested by researchers are also described.

6.2 Previous work by WGECO to define targets, thresholds and limit reference points

6.2.1 Relevant WGECO work

WGECO has carried out and reported a significant body of work relevant to the MSFD and its eleven descriptors (Annex 1). In particular it has been prominent in developing the Driver Pressure State Impact Response (DPSIR) approach, and identification of indicators and reference levels. It has also been prominent in developing the Integrated Ecosystem Assessment approach used in the OSPAR QSR process in 2010, often referred to as the Robinson *et al*. approach. The specific issue of fishery impacts on the ecosystem has also been regularly addressed, and this is of particular importance for the establishment of GES in the context of descriptors 1, 3, 4 and 6. Within this, there has been work on the impacts of different fishing gears, and fishing methods, and guidance on the development of indicators of fishing activity and effort. Specific work was directed at the development of the Large Fish Indicator, now stipulated in the EU Data Collection Framework (DCF) as an indicator of fish community structure and in the MSFD as a key indicator in descriptor 4 on foodwebs.

In WGECO (2010) there is a review of methods to define GES. In this review suitability of methods from other directives, i.e. Water Framework Directive (WFD) (2000/60/EC), the Habitats Directive (HD) (92/43/EEC), are considered and an attempt was made to place the MSFD in the context of these directives. Additional information was collected from these directives that could help defining GES.

Two main topics relevant to previous WGECO work are dealt with below in more detail: (1) the selection of indicators and (2) the setting of reference levels. The way that these topics fit in the overall process is demonstrated in the development of one specific indicator: the Large Fish Indicator (LFI) (Section 6.2.5). These topics are dealt with in the following sections.

6.2.2 Relation to other directives

In WGECO 2010 there is a review of methods used to determine relevant "good status" under the WFD, HD and MSFD, including a discussion of reference points and indicators. For each of these Directives the following was extracted:

- The geographic coverage of the Directive/Guidelines and ways to opt areas in or out;
- The higher level objectives of the Directive/Guidelines and the reference levels that form the objectives;
- The assessment process used (or proposed), the process used to select the indicators, and methods to combine information from the individual indicators into the evaluation of status;
- The approaches used to achieve consistency in the evaluations of status under the Directives/guidelines across regions and subregions.

From this information a way forward is proposed with implementation of the MSFD that is scientifically sound, while being as efficient as practical in demands for monitoring and assessment, and harmonious with implementation of other Directives.

To further elaborate on this work initiated in WGECO (2010) the details as specified in the WFD and HD are given below. The role of the HD and WFD to provide methodological standards for setting reference levels in the MSFD has been further elaborated on the basis of the recent JRC report "Review on methodological standards related to the Marine Strategy Framework Directive criteria on Good Environmental Status (Piha and Zampoukas, 2011).

The authors screened methodological standards (defined in the report as "all methods developed and agreed in the framework of European or international conventions") considering the following sources: WFD (2000/60/EC), EQS Directive (2008/105/EC), Habitats Directive (92/43/EEC), Birds Directive (2009/147/EC), Common Fisheries Policy (CFP), Regional Sea Conventions covering European seas (OSPAR, HELCOM, UNEP MAP, Black Sea Commission; Table 6.2.2.1).

Table 6.2.2.1. Availability of methodological standards by MSFD GES Descriptor. X indicates the existence of at least one standard related to assessment, environmental targets or monitoring (Piha and Zampoukas, 2011).

	WFD	EQS Directive	Habitats Directive	Birds Directive	CFP	Regional Sea Conventions	Other Sources
D1 Biological diversity	X		X		X	X	
D2 Non-indigenous species						X	
D3 Commercial fish					X		X
D4 Food webs	X					X	X
D5 Eutrophication	X					X	X
D6 Sea floor	X		X			X	X
D7 Alteration of hydrographical conditions	X						
D8 Contaminants and pollution effects	X	X				X	
D9 Contaminants in fish and other seafood							X
D10 Litter						X	X
D11 Energy/Noise							X

Piha and Zampoukas (2011) reviewed methodological standards for:

- the assessment of the status of the marine environment and the determination of GES;
- assessment of environmental targets;
- routine monitoring.

The report describes marine regions where at least one methodological standard for defining "environmental targets" is available i) for the whole area, ii) only part of the marine region or iii) where no standards are currently available.

A synthesis of this analysis (Table 6.2.2.2) shows the presence of many gaps in the availability of methodological standards for assessing "environmental targets" in all the Descriptors 1, 3, 4, 6. 'Environmental target', according to Article 3, being "*a qualitative or quantitative statement on the desired condition of the different components of, and pressures and impacts on, marine waters in respect of each marine region or subregion*" (see also Section 1.6.1).

Table 6.2.2.2. Availability of methodological standards for "environmental targets" in relation with MSFD GES Descriptors 1, 3, 4, 6 and related indicators, according to main marine regions (NEA: Northeast Atlantic, .BAL: Baltic Sea, MED: Mediterranean Sea, BS: Black Sea). Green: at least one methodological standard available for all of the region; Yellow: at least one methodological standard available only for part of the marine region; Red: no standards currently available (Piha and Zampoukas, 2011; modified).

Descriptor/Indicator	Indicator	NEA	BAL	MED	BS
1. Biological Diversity					
Species distributional range	1.1.1				
Distributional range of habitat	1.4.1				
Habitat area	1.5.1				
Area covered by the species (for sessile/benthic species)	1.1.3				
Population abundance and/or biomass, as appropriate	1.2.1.				
Population demographic characteristics (e.g. body size or age-class structure, sex ratio, fecundity rates, survival/mortality rates)	1.3.1				
Distributional pattern of habitat	1.4.2				
Condition of the typical species and communities of the habitat	1.6.1				
Relative abundance and/or biomass of the habitat, as appropriate	1.6.2				
Physical, hydrological and chemical conditions of the habitat	1.6.3				
Composition and relative proportions of ecosystem components (habitats and species)	1.7.1.				
Distributional pattern within their distributional range, where appropriate	1.1.2				
Population genetic structure, where appropriate	1.3.2				
Habitat volume, where relevant	1.5.2				
3. Commercial fish					
Available standards for fishing mortality (F)	3.1.1				
Spawning-stock biomass	3.2.1				
Catch/biomass ratio	3.1.2				
Biomass indices	3.2.2				
Proportion of fish larger than the mean size of first sexual maturation	3.3.1				
Mean maximum length across all species found in research vessel surveys	3.3.2				
95% percentile of the fish length distribution in research vessel surveys	3.3.3				
Size at first sexual maturation, which may reflect the extent of undesirable genetic effects of exploitation	3.3.4				
4. Foodwebs					
Performance of key predator species using their production per unit biomass (productivity)	4.1.1				
Large fish (by weight)	4.2.1				
Abundance trends of functionally important selected groups/species	4.3.1				
6. Sea Floor					

Descriptor/Indicator	Indicator	NEA	BAL	MED	BS
Type, abundance, biomass and areal extent of relevant biogenic substrata	6.1.1				
Presence of particularly sensitive and/or tolerant species	6.2.1.				
Multimetric indices assessing benthic community condition and functionality	6.2.2				
Extent of the seabed significantly affected by human activities for the different substrata types	6.1.2				
Biomass or number of individuals in the macrobenthos above some specified length/size	6.2.3				
Parameters describing the characteristics (shape, slope and intercept) of the size spectrum of the benthic community	6.2.4				

A general introduction to the HD has been provided in the WGECO 2010 report (Section 6.2.2). This Directive is mainly relevant to MSFD when setting reference targets and limits in the context of Descriptors 1 (Biological Diversity) and 6 (Sea floor) especially regarding species' abundance, distribution and habitats (ANNEX 2). However, reference levels are currently available only for those species and habitats listed in Annex I and Annexes II, IV, V of the Directive.

For the WFD, the setting of the ecological standards and reference levels, the quality elements to be taken into account for the assessment of water status, and the intercalibration of results over Geographical Intercalibration Groups have been already reviewed by WGECO in 2010 (Section 6.2.1).

According to Piha and Zampoukas (2011) the definition of thresholds/limits for GES indicators may benefit from work conducted for transitional waters and coastal areas (up to one mile from the coastline) in different Member States in the context of the WFD (ANNEX 2).

6.2.3 Selection of indicators

Past work by WGECO is presented in ICES Cooperative Research Report No. 272 (ICES, 2005), where an extensive overview is given of the conceptual background and practical implementation of ecological quality objectives, reference points and fishing effects. This is effectively the definitive background document for the development of the OSPAR/ICES Ecological Quality Objectives (EcoQOs). The report further elaborates metrics for evaluating the ecosystem effects of fishing including databased community metrics, mass-balance models and changes in life history. This covers work done by WGECO between 1998 and 2001.

Building on that work, Section 6.2 in WGECO 2005 is devoted to the selection of indicators distinguishing different kinds of indicators, e.g. "surveillance" vs. "performance" (WGECO 2004) indicators and how this relates to the application of (Rice and Rochet, 2005) selection criteria for Pressure, State, Response indicators and possible approaches to develop those.

One crucial quality that an indicator requires for it to be useful in a management context is the signal to noise ratio. Managers risk losing credibility and resources if they respond to noise rather than to signal (Rice, 2003). For this reason it is essential to have accurate information on the time required to detect changes in indicators, the strength of signal associated with a change in indicator value, or to account for error when setting Targets and Limits. Information on power to detect change and the extent to which indicators are informative can be achieved by signal detection theory and power analysis (ICES 2003, 2004b; Rice, 2003, Nicholson and Jennings, 2004; Piet and Rice, 2005).

Finally WGECO (2010, Section 3.5 on integrated assessments) provide specific information on the selection of indicators in order to do an initial assessment.

6.2.4 Setting reference levels

According to WGECO (2010), once decisions have been made about the indicators that will be used in each national or regional assessment, the first thing to do is to identify reference levels for each indicator; these can be based on both pristine conditions as well as sustainable exploitation.

In order to set a reference level for pristine conditions it is necessary to have some idea of what state the indicator would have been in, at a time where human activities were not impacting the parts of the ecosystem measured by the indicator. This can be identified based on a time (or area) without human impacts or if this information is lacking then some scientifically sound method will be required to project backwards what value the indicator would have had prior to human impacts. This can be done through process-based models supported by adequate data (e.g. Jennings and Blanchard, 2004).

When setting reference levels for an indicator that reflect “sustainable use”, it is necessary to apply a line of consistent ecological reasoning regarding the level of alteration that is *not* sustainable, then set reference levels to avoid this. As summarized in past WGECO reports, there has been substantial scientific debate about appropriate benchmarks for the boundary between sustainable and unsustainable use, and the appropriate ways to deal with uncertainty and natural variation in this boundary condition (ICES, 2006; 2008). The reasoning was developed most fully in building the fisheries advisory frameworks, and although ICES is changing that framework to accommodate a new EU policy objective for fisheries, this rationale remains a useful guide to setting reference levels associated with sustainable use. Additional guidance on standards that are being accepted by both science and policy communities as benchmarks for sustainability can be found in the work of higher-level intergovernmental marine agencies such as FAO and CBD (FAO, 2008; CBD, 2008). Although the reasoning applied in the fisheries advisory frameworks was developed for populations, WGECO has already argued that the same general approach can be followed to estimate reference levels associated with impairment of capacity to recover for other ecosystem attributes (ICES, 2008). Rice (2009) described methods to make the approach operational.

6.2.5 Specific example: Large Fish Indicator

The Large Fish Indicator (LFI) was proposed as an indicator for the state of foodwebs by the ICES/JRC task group dealing with descriptor 4. It is also being used within the CFP to help support the ecosystem approach to fisheries management. This indicator has been developed by WGECO and WGFE over several years (2006–present. The process is described in detail by Greenstreet *et al*. (2011) and, again, focuses on two aspects that need to be considered in order to assess whether the Ecological Quality Objective (EcoQO) for the fish community is achieved:

- Indicator selection, i.e. which metric would be the best state indicator; and
- Setting of target level, i.e. how might the management target, the EcoQO, be set for this indicator.

In order to select the best indicator, a suite of potential fish community indicators reflecting both the species composition and the size distribution were evaluated against ICES (2001) criteria (later developed into the Rice and Rochet (2005) criteria) and the LFI emerged as the best option. After selection, this indicator was further developed to improve its signal to noise ratio. This was achieved by setting the "large fish" threshold as >40 cm (rather than 30 cm) and determining the indicator based on the proportion of the total fish biomass, rather than numbers, exceeding this threshold. This excluded much of the noise caused by stochastic recruitment-driven variation in the juvenile fish. Then a reference level was established based on historical levels when exploitation was considered to be sustainable and the target level was set accordingly. Further detail of the method can be found in the review of the LFI in ToR a.

6.3 Approach by Member States to Descriptors 1, 3, 4, 6

6.3.1 Approach by the United Kingdom

The UK is taking forward the determination of GES in a common programme, in order to support a cost–benefit analysis that will form part of the Initial Assessment due in 2012. The emphasis on potential costs of implementation has been used to guide national activity in relation to pressure descriptors, and to a lesser extent for the biodiversity descriptors identified above. Separate discussions have been undertaken to plan integrated monitoring programmes, but this is at an early stage and little has been agreed in this area. Trans-boundary development of descriptor 3 (fish and shellfish) by ICES will provide valuable support to the UK approach.

6.3.1.1 Monitoring needs to support indicators

Initial assessment: UK approach

An initial assessment of many ecological characteristics that could be used to describe biodiversity descriptors has been undertaken by the UK (Charting Progress 2, 2010; Defra). This comprehensive report describes the current status and trends of predominant habitats and species characteristic of UK territorial waters, and it is expected that much of these data will be sufficient to inform whether GES is met in the regional sea. The supporting data will require careful evaluation for applicability and relevance to the MSFD and GES but it is assumed that current GES assessments will be supported by existing monitoring programmes.

Existing monitoring programmes

There are a number of monitoring programmes in place that should meet many of the requirements of the MSFD. The programmes are detailed within several sources in the UK and elsewhere, including:

- European Directory of the Ocean-observing System (EDIOS);
- United Kingdom Directory of the Marine-observing Systems (UKDMOS);
- International Council for the Exploration of the Sea (ICES);
- European Marine Observation and Data Network (EMODNET); and
- Mapping European Seabed Habitats (MESH).

Put together, these sources provide a comprehensive overview of existing monitoring practices in the UK. The programmes have been compiled from data covering a wide range of ecosystem characteristics, often over many years. Many monitoring programmes were developed to support existing legislation (e.g. Common Fisheries Policy, Habitats Directive and Water Framework Directive), each with their own objectives and approaches for monitoring but which are consistently applied throughout multiple European regions (e.g. OSPAR Commission, 2009; Topcu *et al*., 2009; Backer *et al*., 2010; HELCOM, 2010).

The UK has yet to develop new monitoring programmes for MSFD but the process to select specific indicators and develop targets to evaluate the outputs of the monitoring programmes in terms of GES is underway.

6.3.1.2 Developing indicators

An important step in the UK process for developing indicators and targets for the MSFD was an expert workshop on "Marine Biological Targets and Indicators for the Marine Strategy Framework Directive", held on 29–31 March 2011 in Birmingham. The purpose of the workshop was to define indicators and targets for MSFD Descriptors 1 (biological diversity), 4 (foodwebs), and 6 (sea floor integrity).

The UK has established six working groups addressing different components of the marine ecosystem: seabirds, marine mammals and turtles, sediment habitats, rock and biogenic reef habitats, fish and cephalopods, and pelagic habitats. Work within each working group was undertaken to provide sets of suitable indicators for the four biodiversity descriptors. A review process will establish whether there is a sufficiently complete set of indicators for each descriptor or whether further work is required.

Discussion at the workshop focused on identifying those indicators for Descriptors 1, 4, and 6 stipulated in the EC Decision documents that could be populated using data available from current monitoring programmes. Thus for example, the fish subgroup considered that indicators related to distribution range, distributional pattern, abundance, biomass, population demographic condition, and composition and relative proportion could all be readily developed and used to assess status in respect of Descriptor 1 (Biodiversity). The discussion then focused on selecting the most appropriate species on which to base these indicators then proposing targets for these indicators both at the individual species indicator level and at the criterion level.

It was agreed that some metrics, such as the Large Fish Indicator and indicators on the productivity of key predator or prey species, could meet the needs of the indicators prescribed by the Commission Decision document (2010/477/EU). However, it was not clear that these indicators would alone convey sufficient information regarding marine foodwebs to ensure that GES could be achieved. The EC Decision document recognizes the fact that indicators for the foodweb Descriptor need further development; this need was endorsed by the workshop's participants (see also discussion in ToR a (LFI)).

6.3.1.3 Methods used to identify thresholds and reference levels

The scientific understanding underlying some indices specified by the Commission Decision document (2010/477/EU) may be insufficient for setting targets even when good baseline data are available. For example, the interpretation of criterion 4.1 "Productivity (production per unit biomass) of key species or trophic groups" depends on whether population abundances are limited by food or by other ecological constraints. In the former case, one would expect to see low productivity for a population near carrying capacity (a desirable outcome). In the latter case, high productivity could be indicative of a high potential or actual carrying capacity. The dominating mechanism limiting carrying capacity of marine populations is unknown in many cases.

6.3.1.4 Linking to management measures

UK is gathering information on potential management measures in order to inform the economic analysis of these measures (as required under Article 13.1 of the Directive), and to identify if additional measures are needed in order to reach and maintain GES. At a workshop held in early 2011, experts in marine science, policy and economics met to identify the components of a Cost–benefit Analysis (CBA) that is required under the Directive. Specifically this has three stages; 1) to predict the status of the UK marine environment in 2020 under a business-as-usual (BAU) scenario, 2) to assess the effectiveness of potential management measures in alleviating pressures on the marine environment, and 3) to identify the potential costs of implementing these management measures and the stakeholders affected.

The work is expected to quantify short-term costs and benefits, and also to provide a first understanding of longer term costs and benefits to inform the 2011 consultation on targets. The work should allow systematic comparison of the different options to reduce pressures on the ecosystem in order to ensure that GES is achieved or maintained.

Potential management measures have been grouped according to the type of pressure on the marine environment they aim to alleviate or eliminate, e.g:

- physical damage to the seabed;
- other physical disturbance (litter, noise);
- interference with hydrological processes (turbidity, barriers);
- adverse by-products of human activity (nutrients, contaminants);
- biological disturbance (extraction, non-native introductions).

The measures were also scored according to their effectiveness in alleviating pressures, rather than their effectiveness in helping to achieve GES. This is because the definition of GES in UK waters has not yet been finalized and may even then be subject to changes over time, and because it is through the alleviation of pressures, among other things, that GES can be achieved or maintained.

6.3.1.5 Assessment of cost of degradation

Article 8.1 of the Marine Strategy Framework Directive requires that the cost of degradation of the marine environment be estimated; in other words, the value of the loss, if any, of ecosystem goods and services should no further policy action be taken. The BAU scenario that was established in the workshop will be used as the baseline in analysing the cost of degradation.

This analysis will use the following assumptions:

- Anticipated changes to ecosystem components and characteristics by 2020;
- A future use of the marine environment that takes into account economic growth and continued implementation of existing policies;
- Existing management measures continue to be implemented if feasible (e.g. those which have been implemented under the Water Framework Directive);
- New non-MSFD measures are implemented as planned (e.g. a network of Marine Protected Areas will be in place);

- Sectoral initiatives that have been licensed are implemented as planned (e.g. offshore renewables).

Climate change is an underlying factor causing change in natural conditions, and will therefore be treated as a separate driver which causes changes in ecosystem components and characteristics. The analysis will attempt to keep separate the degradation caused by immediate human and sectoral activities from those caused by climate change as the former can be managed through relatively local measures whereas climate change has to be addressed at the global level.

6.3.2 Approach by Germany to Descriptors 1, 3, 4, 6

6.3.2.1 Monitoring needs to support indicators

Germany has not finished selecting indicators and defining targets for GES. The initial assessment is almost completed and draws upon existing assessments under the WFD, HD, HELCOM, OSPAR, and ICES. The impact of generic pressures was evaluated for different ecosystem components including fish, birds, mammals, etc. by using the results of existing monitoring programmes.

The government and federal states conduct coordinated long-term monitoring programmes within the coastal and offshore waters. Some of the observations are of direct use in assessing the state of indicators under descriptors 1, 3, 4 and 6. In Table 6.3.2.1.1 the relevant contents of the programmes are related to the descriptors.

Table 6.3.2.1.1. German monitoring programmes that are of use for the indicator assessment under descriptors 1, 3, 4 and 6.

Descriptor	Monitoring specification
1	Macrophytes (coastal)
	Macrozoobenthos
	Morphology, Substrata
	Phytoplankton
	Zooplankton (local, meso- and macrozooplankton)
	Birds (seabirds and coastal birds, Seabirds-at-Sea programme)
	Fish (coastal and offshore)
	Mammals
	(under development: Habitat mapping activities)
3	ICES assessments
	Fish (offshore)
	VMS data collection
4	Macrophytes (coastal)
	Macrozoobenthos
	Morphology, Substrata
	Phytoplankton
	Zooplankton (local, meso- and macrozooplankton)
	Birds (seabirds and coastal birds, Seabirds-at-Sea)
	Fish (coastal and offshore)
	Mammals
6	(under development: Habitat mapping)
	Macrozoobenthos
	Morphology, Substrata
	VMS data collection

6.3.2.2 Developing indicators

Multiple groups combining experts from numerous agencies and institutions representing government and federal states are working on the MSFD implementation process in Germany. These groups have yet to finalize any definitions of GES.

Initial definitions of the terminology used under the MSFD have been presented to OSPAR (OSPAR GES4BIO Doc. 2). Germany will probably adopt the OSPAR approach in defining indicators for the respective descriptors. This approach considers four scales of ecosystem component (species, functional groups, habitat types, ecosystems), which may be subdivided into a number of key elements (e.g. individual species).

a) Species-individual species, such as those listed under Community Directives or identified as key species for assessment of a wider functional group;
b) Functional groups-covering the seabirds, mammals, reptiles and fish and representing the main functional groups of more mobile taxa;
c) Habitat types-predominant and special (listed) types, covering both the seabed and water column habitats, and including their associated biologi-

cal communities (in the sense of the term biotope as given in the COM Decision);

d) Ecosystems-where assessment of multiple habitats and functional groups as part of larger ecosystems is envisaged.

These components will be used for the selection of indicators. Each indicator will be applied to a selection of key elements from the above list.

Table 6.3.2.2.1 lists the indicators under the descriptors 1, 3, 4 and 6 which have available monitoring data and have established assessment methods. The 14 indicators will be applied to a selection of key components resulting in a possible list of 14 indicators.

Table 6.3.2.2.1. Criteria and selected indicators for the descriptors 1, 3, 4 and 6.

Criteria for descriptors
1
1.1 Species or functional groups distribution
Distributional range, where appropriate (1.1.1),
Distributional pattern within the latter, where appropriate (1.1.2)
Area covered by the species (for sessile/benthic species) (1.1.3)
1.2 Population size
Population abundance and/or biomass, as appropriate (1.2.1)
3
3.1 Level of pressure of the fishing activity
Fishing mortality (F)(3.1.1)
Ratio between catch and a biomass index (hereinafter catch/biomass ratio) (3.1.2)
3.2 Reproductive capacity of the stock
Spawning-stock biomass (SSB) (3.2.1)
3.3 Population age and size distribution.
Proportion of fish larger than the size of first sexual maturity (3.3.1)
4
4.2 Proportion of selected species at the top of foodwebs
Large fish (by weight) (4.2.1)
4.3 Abundance/distribution of key groups/species
Abundance trends of functionally important selected groups/species (4.3.1)
6
6.1 Physical damage, having regard to substrata characteristics
Type, abundance, biomass and areal extent of biogenic substrata (6.1.1)
Extent of the seabed affected by human activities for the different substrata types and frequency of disturbance (e.g. bottom trawls per year)(6.1.2)
6.2 Condition of benthic community
Presence of particularly sensitive and/or tolerant species (6.2.1)
6.3. Oxygen concentrations in bottom water and / or upper sediment layer
Extent of area with spatial and temporal hypoxia

6.3.2.3 Methods used to identify thresholds and reference levels

Germany will align its work to implement MSFD with that of existing directives such as the WFD and HD. This includes an alignment of assessment methodologies. In addition, recommendations and assessment approaches from OSPAR and HELCOM such as EcoQO's will be considered.

No standard method has so far been defined for the identification of thresholds and reference levels. However, it is likely that for descriptor 3, ICES recommendations will be used to define GES and related threshold and reference levels.

6.3.2.4 Linking to management measures

Linking the indicators and related GES thresholds to management measures is still to be done in the national implementation process. However, the definition of targets and GES will be linked to manageable human activities.

6.4 Approach by researchers to Descriptors 1, 3, 4, 6

6.4.1 Approach of Borja *et al.*

Borja *et al.* (2010; 2011) carried out a study focused on the southern part of the Bay of Biscay. The availability of data in this region is good, and an integrated assessment for environmental status was undertaken. Despite this assessment being made at a smaller scale than the level of ecoregions or subregions (in this case the Bay of Biscay and Iberian coasts), this study aimed to identify and discuss the practical problems in implementing the MSFD at a broader scale and to guide future assessments.

6.4.1.1 Monitoring needs to support indicators

Various monitoring programmes have been undertaken in the Basque Country in the past 25–30 years in the coastal and offshore marine waters within the framework of European, national and regional projects (Borja *et al*., 2011) in support of existing legislative requirements. In particular, extensive work has been carried out to fulfil the requirements of the Water Framework Directive, the Habitats Directive and the Common Fisheries Policy. Some of this information permits the assessment of some of the biodiversity indicators (marked with an asterisk in Table 6.4.1.1.1).

Table 6.4.1.1.1 shows the indicators that have been used in this case study for Descriptors 1, 3, 4 and 6 of the MSFD. An Ecological Quality Ratio (EQR) was used to define the environmental status of the area using information derived from each of these indicators (Borja *et al.*, 2011).

In addition to the need to identify indicators and set appropriate targets, there will also be a need to combine indicators to assess whether overall GES is being achieved. Borja *et al.*(2010) describe one way of carrying out this integrated assessment. These authors propose a method to combine indicators by grouping marine ecosystem components into four distinct (but interlinked) systems: (i) water and sediment physico-chemical quality (including general conditions and contaminants); (ii) planktonic (phyto- and zooplankton); (iii) mobile species (fishes, sea mammals, seabirds, etc.) and (iv) benthic species and habitats. These ecosystem components, affected by different human pressures, can be related to the 11 qualitative descriptors and, as such, to different indicators able to provide information on the quality of all these elements.

6.4.1.2 Methods used to identify thresholds and reference levels

The methods used to identify thresholds and reference levels in the Borja *et al*. (2011) approach are varied and depend on the characteristics of each descriptor. A summary of methods used for each descriptor is given below. The final EQR of each of these descriptors and its reliability is shown in Table 6.4.1.2.2, whereas the total EQR and total reliability (together with the EQRs for the other descriptors of the MSFD) can be seen in Borja *et al*., 2011; Table 8).

a. Biodiversity

The complexity of this descriptor makes it difficult to integrate all the available information. However, Borja *et al*. (2011) use a biodiversity valuation approach as an integrative tool to assess biodiversity based on zooplankton, macroalgae, macroinvertebrates, demersal fishes, sea mammals and seabirds data collected from 2003 to 2009 (see Figure 2 *in* Borja *et al*., 2011).

It is possible to integrate the biodiversity valuation into a unique value for the whole study area (see details of this approach in Borja *et al.* (2009)). In this case, and due to the lack of reference points for this descriptor, the 'high' values obtained in the biodiversity valuation will be used as environmental targets (see also Borja *et al*. (submitted)).

b. Exploited fish and shellfish

Fishing mortality, spawning-stock biomass and population age and size distributions have been used as population indicators for all commercial fish and shellfish. The most relevant stocks were analysed (twelve stocks in total; see Table 5 in Borja *et al.* (2011)). Reference fishing mortality values were available for most of them.

c. Marine foodwebs

The approach proposed for this descriptor is limited by the fact that regular monitoring of the lower trophic levels is limited to the coastal area, but exploited fish over the wider area. In this case, hake productivity has been selected as an indicator of changes in the productivity for a key predator (see Borja *et al*. (2011) for further details).

In relation to the proportion of large fish in the study area, data from the French bottom-trawl survey (EVHOE, IBTS framework) have been used. Although these data are not restricted to the waters of the study area, they can be considered to be representative of this area, considering the whole of the Bay of Biscay as a continuum. The data have been compared to the North Sea IBTS results.

Trends in the abundance/biomass of functional groups, such as phytoplankton, mesozooplankton, sardine and anchovy (as small pelagics), horse mackerel and mackerel (as intermediate pelagics), and hake (as demersal fish) have been considered, using ICES data. However, it is necessary to bear in mind that different stocks are evaluated at different scales and it is difficult to extrapolate variations to a particular regional scale.

d. Seafloor integrity

The extent of the seabed affected significantly by human activities represents 2.3% of the study area. This impact can be considered as moderate, when assessing the benthic ecological status using the index M-AMBI: multivariate AMBI - AZTI's Marine

Biotic Index (Borja and Collins, 2004; Muxika *et al.*, 2007). The rest of the area presents a good quality status (see Borja *et al.*, 2011 for further information).

6.4.1.3 Linking to management measures

The assessment of the environmental status in the Bay of Biscay and the calculation of an EQR for the descriptors of the MSFD (and their indicators; see Table 6.4.1.2.3 in this document and Table 8 in Borja *et al.*, 2011) permits the development of management plans for reducing the human pressures (mainly fishing) that adversely affect the descriptors, at levels than can improve the EQR each of the indicators and descriptors used in the assessment.

Table 6.4.1.2.2. Qualitative descriptors and different aspects and indicators to be used in the environmental status assessment, selected by the European Commission (2010). (from Borja *et al.*, 2011, Table 2). Asterisks show the indicators used in this assessment. (See Borja *et al.*, 2011 for further explanations).

DESCRIPTOR	ASPECT	INDICATOR
1: Biological diversity	1.1 Species distribution	1.1.1 Distributional range*
		1.1.2 Distributional pattern within the latter*
		1.1.3 Area covered by the species (for sessile/benthic species)*
	1.2 Population size	1.2.1 Population abundance and/or biomass*
	1.3 Population condition	1.3.1 Population demographic characteristics
		1.3.2 Population genetic structure
	1.4 Habitat distribution	1.4.1 Distributional range*
		1.4.2 Distributional pattern*
	1.5 Habitat extent	1.5.1 Habitat area*
		1.5.2 Habitat volume, where relevant
	1.6 Habitat condition	1.6.1 Condition of the typical species and communities
		1.6.2 Relative abundance and/or biomass, as appropriate*
		1.6.3 Physical, hydrological and chemical conditions
	1.7 Ecosystem structure	1.7.1 Composition and relative proportions of ecosystem components (habitats, species)*
3: Exploited fish and shellfish	3.1 Level of pressure of the fishing activity	3.1.1 Fishing mortality (F)*
		3.1.2 Catch/biomass ratio
	3.2 Reproductive capacity of the stock	3.2.1 Spawning-stock biomass (SSB)*
		3.2.2 Biomass indices
	3.3 Population age and size distribution	3.3.1 Proportion of fish larger than the mean size of first sexual maturation*
		3.3.2 Mean maximum length across all species found in research vessel surveys
		3.3.3 95 % percentile of the fish length distribution observed in research vessel surveys
		3.3.4 Size at first sexual maturation*
4: Foodwebs	4.1 Productivity of key species or trophic groups	4.1.1 Performance of key predator species using their production per unit biomass*
	4.2 Proportion of selected species at the top of foodwebs	4.2.1 Large fish (by weight)*

DESCRIPTOR	ASPECT	INDICATOR
	4.3 Abundance/distribution of key trophic groups/species	4.3.1 Abundance trends of functionally important selected groups/species*
6: Seafloor integrity	6.1 Physical damage, having regard to substrata characteristics	6.1.1 Type, abundance, biomass and areal extent of relevant biogenic substrata*
		6.1.2 Extent of the seabed significantly affected by human activities for the different substrata types*
	6.2 Condition of benthic community	6.2.1 Presence of particularly sensitive and/or tolerant species*
		6.2.2 Multimetric indices assessing benthic community condition and functionality, such as species diversity and richness, proportion of opportunistic to sensitive species*
		6.2.3 Proportion of biomass or number of individuals in the macrobenthos above specified length/size
		6.2.4 Parameters describing the characteristics of the size spectrum of the benthic community

Table 6.4.1.2.3. Assessment of the environmental status, within the Marine Strategy Framework Directive, in the Basque Country offshore waters (Bay of Biscay), taking into account the eleven qualitative descriptors and the indicators used for the assessment (see Table 6.4.1.2.2, for listing).

Qualitative Descriptors	Indicators used	Explanation	Reference conditions/EQS	Recent trend	Reliability (%)	Weight (%)	EQR
1.- Biological Diversity	111, 113, 121, 141, 142, 151	integrated biological value		NA	69	15	0.51
3.- Exploited fish and shellfish				▼	100	15	0.69
	311	fishing mortality <reference			100		0.82
	321	Spawning stock <reference			100		0.67
	331	% large fish			100		0.59
4.- Marine foodwebs	411, 421			▼	70	10	0.40
6.- Seafloor integrity			WFD	▶	100	10	0.89
	612	Area not affected			100		0.87
	621	% presence sensitive sp.			100		0.98
	622	Mean M-AMBI value			100		0.83

6.4.2 Case study of Options for Delivering Ecosystem-based Marine Management (ODEMM), an FP7 project

The FP7 project ODEMM aims to develop a framework of options for delivering ecosystem-based marine management to support the objectives of the MSFD. These options will include operational objectives, be fully costed, and be in language that is readily understood by managers and stakeholders.

The European Commission has provided guidance on the criteria and methodologies for evaluating the 11 GES descriptors (Commission Decision document 2010/477/EU). Many Member States intend to use existing assessments (e.g. Charting Progress 2) to complete the initial assessment, but this is not straightforward. For example, the Habitats Directive (92/43/EEC) provides a common status assessment approach for the Baltic, NE Atlantic and parts of the Mediterranean Sea (i.e. Favourable Conservation Status of listed species and habitats). However, the baseline and specific objectives of the assessment are different from those that may be used by the MSFD. Other regionally specific assessments may have a differing basis for assessment, objectives and criteria (e.g. OSPAR's Quality Status Report in the NE Atlantic, the Black Sea's Status of the Environment reports).

6.4.2.1 Risks to achieving GES

The ODEMM project has developed a risk assessment approach (Breen *et al.*, in prep.) using existing environmental status and trend assessments. The purpose is to identify the extent of departure of current biodiversity status from GES, and therefore the scale of the current risk and effort required to achieve GES. The assessment compares current status against three risk criteria for GES based on ODEMMs interpretation of the GES definitions.

The following sections describe the ODEMM interpretation of GES, and three risk criteria (high, medium and low) describing increasingly degraded states of ecological characteristics under each Descriptors 1, 3, 4 and 6. Risk category criteria are considered in order starting at "high risk". The "and/or" statement determines whether one or multiple criteria are used in evaluation of risk categories. If the criteria for high risk are not satisfied then the next lowest i.e. moderate risk criteria are then evaluated.

6.4.2.1.1 MSFD Descriptor 1: Biodiversity

Under Annex I of the MSFD (2008/56/EC), the qualitative descriptor for determining biodiversity GES is defined as 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 climatic conditions'. ODEMM considers this a clear definition of how GES will be evaluated, specifically that <u>biological diversity is maintained</u> i.e. no further loss, in line with prevailing conditions. However, ODEMM recognizes that this definition is in conflict with the requirement stipulated in Article 3(5) of the directive defining GES as 'the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable.' This definition explicitly requires the ecosystem components i.e. species and habitats, to be in sustainable condition and does not allow for the situation where an ecosystem component is currently in unfavourable (unsustainable) condition. In such cases, if 'no further loss' of biodiversity is achieved, then under the Annex I definition, GES is achieved. Thus the risk assessment framework does not address a scenario where biodiversity is below GES. However, the same approach could be easily adapted to work in this situation.

ODEMM considers that Good Environmental Status is achieved when biodiversity is maintained i.e. no further loss, in the regional sea such that the quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions. Following the definition provided in Annex I (2008/56/EC), failure to achieve GES will occur when there is loss of biodiversity beyond that expected under prevailing conditions before 2020. Loss can be described as a reduction in genetic, species, habitat or ecosystem diversity within the regional sea and severe examples would include extirpation of meta-populations, species, habitat types or ecosystem properties within a regional sea. Biodiversity loss can also be considered as a noticeable change in diversity resulting from a shift in dominance or change in species evenness. However, in both cases the degree of loss would need to exceed levels expected under prevailing conditions.

This definition recognizes that the prevailing conditions are those that currently exist, and which incorporate the effects of sustainable exploitation on the physical and biological components of the ecosystem.

The risk assessment has been developed with three tiers of risk: (1) High, (2) Moderate, and (3) Low (Table 1.4.2.1.2.1). The approach assigns both a risk category and a confidence level for each Ecological characteristic evaluated and uses current status and trend information (in this case describing NE Atlantic species and habitats) to determine the category of risk assessment and the level of confidence in this assessment. The level of confidence was also assigned using a three-tier assessment (High, Moderate or Low) and determined using a series of criteria evaluating the quality of the background data, the interpretation of that data and agreement among experts.

For biodiversity, the risk categories are influenced by the perception of current status as defined under existing status assessment frameworks e.g. Habitats Directive, Water Framework Directive. The risk assessment provides an initial evaluation of the level of departure from 'good' status of an ecosystem component within existing frameworks. The criteria for 'good' status in some status frameworks are likely to be more stringent (e.g. HD, WFD) than required for the MSFD. Therefore, assessment of risk using the criteria below (shown in Tables 6.4.2.1.2.1 to 6.4.2.1.8.1) provides an indication of the likelihood of achieving good status only under the MSFD.

6.4.2.1.2 Assessing GES for Biodiversity

Commission Decision document (2010/477/EU) recommended that GES for Biodiversity should be assessed individually for major ecosystem characteristics listed in Annex III of the MSFD and guidance on the criteria and indicators provided. This will require multiple species and habitat indicators to be assessed using seven criteria describing population and habitat parameters. In broad terms, GES can be interpreted as being achieved when the indicators (and criteria) are (i) in good status under existing assessments (e.g. Favourable Conservation Status under the Habitats Directive), and/or (ii) demonstrate a stable or increasing trend. Some Member States are beginning to investigate how multiple species or habitats will be combined within a single estimate of GES.

Table 6.4.2.1.2.1. Three risk categories for assessing failure to achieve GES for Biodiversity (after Breen *et al.*, in prep).

High	Loss of a genotype, species, habitat or ecosystem type at the regional scale (decline in biodiversity) within the next 10 years (= extirpation) and/or Maintained change in the dominance of genotypes, species, habitat types or ecosystem types (change in evenness) where this change is likely to last for at least the next 10 years
Moderate	Decline in extent and/or condition of genotypes, species, habitat types or ecosystem types at the regional scale within the next 10 years and/or Alterations in the dominance of genotypes, species, habitat types or ecosystem types (change in evenness) within the next 10 years, not necessarily having led to a maintained change
Low	No notable changes in extent and condition of genotypes, species, habitat types or ecosystems at the scale of the region beyond that expected given prevailing conditions within the next 10 years and No clear change in dominance of genotypes, species, habitat types or ecosystem types (change in evenness) given prevailing conditions within the next 10 years

6.4.2.1.3 MSFD Descriptor 3: Commercial fish and shellfish

ODEMM considers that GES for commercially exploited fish and shellfish can be described as a status where stocks are sustainably exploited consistently with high long-term yields and have full reproductive capacity. It will also be necessary for the age and size distribution of fish and shellfish populations to be representative of a healthy stock.

6.4.2.1.4 Assessing GES of Commercial Fish and Shellfish

The Commission recommended assessment of status using three criteria describing 1) levels of fishing pressure, 2) reproductive capacity and, 3) population size and distribution. Three primary indicators are suggested namely, fishing mortality (F), spawning-stock biomass (SSB) and, proportion of large fish. GES would be achieved for a particular stock only if the criteria are fulfilled for all attributes (species).

Table 6.4.2.1.4.1. Three risk categories for assessing failure to achieve GES for Commercial Fish and Shellfish species (after Breen *et al.*, in prep).

High	SSB < SSBpa for some stocks or exploitation rate F exceeds precautionary levels for some stocks (>25%) or the age and size distribution of fish and shellfish stocks shows consistent long-term degradation. i.e. smaller, younger fish.
Moderate	25% stocks are exploited sustainably ($F<F_{MSY}$) and all stocks $SSB > SSB_{pa}$
Low	All stocks are exploited sustainably ($F<F_{MSY}$) or $SSB > SSB_{MSY}$ for >50% of stocks or all stocks $SSB > SSB_{pa}$ or the age and size distribution of fish and shellfish stocks show no degradation. i.e. smaller, younger fish.

6.4.2.1.5 MSFD Descriptor 4: Foodwebs

The interactions between species in a foodweb are complex and constantly changing, making it difficult to identify any one particular condition that represents 'good' status. However, some changes in the relative abundance of species in an ecosystem can have significant adverse effects on foodweb status.

6.4.2.1.6 Assessing GES of Foodwebs

ODEMM considers Good Environmental Status of Foodwebs can be described as a situation where energy flows through the foodweb, and the size, abundance and distribution of key trophic groups/species, are all within acceptable ranges that will secure the long-term viability of all foodweb components in line with prevailing natural conditions. This may be assessed using trends in abundance and distribution of indicator species such as 1° producers or top predators. The commission recommended 3 criteria namely (1) productivity of key species or trophic groups, (2) proportion of selected species at the top of foodwebs, and (3) the abundance and/or distribution of key trophic groups. The criteria used to evaluate the risk of achieving GES are shown below (Table 6.4.2.1.6.1).

Table 6.4.2.1.6.1. Three risk categories for assessing failure to achieve GES for Foodwebs (after Breen *et al.*, in prep).

High	Spatially extensive and long-term changes have occurred in energy flows through the foodweb, as recorded by changes in the productivity (production per unit biomass) of several key species or trophic groups, which have both direct and indirect effects on different trophic levels. or Trends in the abundance and distribution of carefully selected indicator populations, and in the proportion of species at the top of foodwebs, show continuous decline across the Region and provide evidence of adverse impacts on foodweb integrity.
Moderate	Recent changes in the productivity (production per unit biomass) of some key species or trophic groups suggest that direct and indirect effects have occurred on different trophic levels. or Trends in the abundance and distribution of local indicator populations, and in the proportion of species at the top of foodwebs, suggest that adverse impacts to foodweb structure have occurred in some subregions.
Low	Recorded changes in energy flows through the foodweb, as recorded by changes in the productivity (production per unit biomass) of key species or trophic groups, have no significant direct and indirect effects on different trophic levels. or Trends in the abundance and distribution of carefully selected indicator populations, and in the proportion of species at the top of foodwebs, vary in accordance with natural cycles and show no cause for concern in relation to foodweb structure.

6.4.2.1.7 MSFD Descriptor 6: Seafloor Integrity

ODEMM considered that GES for seafloor integrity can be described by a situation where it is at a level that ensures that the structures and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected. The 'seafloor' includes both the physical structure and biotic composition of the benthic community and its 'integrity' includes the characteristic functioning of natural ecosystem processes and its spatial connectedness. The seafloor is considered to not be adversely affected when any impact or pressure that may be occurring does not degrade the natural levels of diversity, productivity, and dynamic ecosystem processes.

6.4.2.1.8 Assessing GES of Seafloor Integrity

Both in the UK and throughout Europe, we have a limited understanding of the structure of benthic habitats. This includes the physical structure (sediment type) and characteristic species of that habitat. This is in part, as a result of the cost and resources required to undertake this type of survey. Therefore, where data are available, they are used to model or predict the seafloor habitat e.g. UKSeaMap. These maps provide broad overviews of the likely habitat type, but may have insufficient detail or certainty to allow a quantitative assessment of GES using target values.

In Charting Progress 2 (2010), the OSPAR approach (Robinson *et al.*, 2008) was used to evaluate seafloor status; an expert judgement approach that follows several steps to evaluate the effect(s) of human activities on ecological characteristics (e.g. predominant habitat type). This approach has been adopted and improved upon within the ODEMM project (Robinson *et al.*, in prep.). The pressure assessment makes no attempt to directly infer the status of the ecological characteristic being evaluated. Simply it provides information on the threat to the characteristic (e.g. a particular predominant habitat such as littoral rock) and the extent of this threat in the regional sea. For some descriptors such as Seafloor Integrity, the outcome of the pressure assessment can represent GES. Under Descriptor 6: Seafloor Integrity, achievement of GES can be interpreted as occurring when there are no widespread severe impacts affecting a predominant habitat type. Here, severe impacts are interpreted as those adversely affecting the characteristic structures and function of the habitat and its typical species. The pressure assessment indicates where pressures (from human activities) overlap with predominant habitat types and when, either solely or in combination, represent severe and widespread impacts (Table 6.4.2.1.8.1).

Table 1.4.2.1.8.1. Three risk categories for assessing failure to achieve GES for Seafloor Integrity (after Breen *et al.*, in prep).

High	Where the pressures and habitats overlap: 1. Extent is widespread (even or patchy), severity is acute or chronic and the persistence of the pressure is high or continuous, irrespective of frequency of occurrence Or 2. Extent is widespread (even or patchy), severity is acute and the frequency of occurrence is occasional or higher, irrespective of Persistence category Or 3. Extent is widespread (even or patchy), severity is chronic and the frequency is persistent or common, irrespective of Persistence category Or 4. A combination of multiple local pressures which result in a widespread extent with a severity, frequency and persistence combination equivalent to one of the above Or 5. The overlap of multiple low severity pressures which combine to form a severe (acute or chronic) impact combination equivalent to one of the above
Moderate	Any combination other than high or low
Low	Where severity is classified as 'low' for all interactions with pressures in the region even when they are combined Or Where any severe effects (chronic or acute) occur and frequency of occurrence is rare, persistence is low, and resilience is high.

6.5 Prioritisation of WGCATDAT Survey Tasks

Under ToR D WGECO was requested to "*Identify elements of the WGECO work that may help determine status for the 11 Descriptors set out in the Commission and provide views on what good environmental status (GES) might be for those descriptors, including methods that could be used to determine status.*"

A key part of this is the need to obtain monitoring data for the various indicators within the descriptors. It is recognized that the bulk of this will likely come from research vessel surveys. In this context WGECO was also asked by the Working Group on Integrating Surveys into the Ecosystem Approach to Fisheries Management (WGISUR) to review the work of the Workshop to catalogue data needs for the EAFM (WKCATDAT). WKCATDAT developed a table of actual or potential data products from fishery surveys, and their relevance to the ten of the eleven Marine Strategy Framework Directive (MSFD) Good Environmental Status (GES) descriptors (Table 6.5.1). No survey tasks were identified which would provide information on the eleventh descriptor, Energy and Noise. The members of WKCATDAT were predominantly data providers and it was felt that input was also needed from data users such as WGECO.

Members of WGECO reviewed this table, and indicated priorities for data collection, provided comments on the spatial and temporal resolution of the data and provided an indication of the immediacy of the data requirement (some tasks could be highly relevant to a research priority but may not be needed immediately to inform indicators for the MSFD descriptors). Scores from 0 (no value) to 3 (high value) were provided for each survey task. The survey tasks were then ranked according to their perceived priority within each of the eight Task Categories (Table 6.5.1).

Table 6.5.1. Proposed Survey Tasks and their Relationship with the MSFD Descriptors[1].

	Scoring Attributes	MSFD Descriptor Related to Task Category												
Task Category	**Total Score**	**N**	**Low Score**	**High Score**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Fish														
Stomach sampling	15	7	0	3	x		x	x						
Additional biological data (e.g. liver/gonad weight, otoliths, scales, fin-rays, length-weight data of other than standard species)	15	7	0	3	x	x	x	x				x		
Genetic information	9	6	1	3	x		x							
Lipid content	8	6	0	2				x						
Tagging	7	4	0	3			x							
Disease/parasite registration	7	5	0	2	x							x	X	

	Scoring Attributes	MSFD Descriptor Related to Task Category												
Task Category	**Total Score**	**N**	**Low Score**	**High Score**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Sonar observations pelagic fish	5	5	1	1			x							
Organism collection (e.g. for contaminants, fatty acids analysis, etc.)	4	6	0	2	x	x	x	x				x	X	
Bioactive materials in marine species	3	4	0	1										
Physical and chemical oceanography [CTD, chlorophyll, oxygen, nutrients, turbidity, etc.]														
Continuous underway measurements	9	3	3	3							x			

	Scoring Attributes	MSFD Descriptor Related to Task Category												
Task Category	**Total Score**	**N**	**Low Score**	**High Score**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Nutrient samples	8	4	1	3					x		x			
Station measurements	6	2	3	3							x			
Autonomic devices	2	1	2	2							x			
Water movement	2	3	0	2							x			
Biological oceanography														
Phytoplankton samples (CPR)	15	6	1	3	x	x	x					x		
Zooplankton samples (towed samples)	15	6	1	3	x	x	x					x		
Phytoplankton samples (water samples)	14	6	1	3	x	x	x		x			x		
Zooplankton samples (dip samples)	11	6	1	3	x	x	x					x		

	Scoring Attributes	MSFD Descriptor Related to Task Category												
Task Category	**Total Score**	**N**	**Low Score**	**High Score**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Zooplankton samples (echosounder)	9	5	1	3	x	x	x					x		
Microbiological samples	7	4	0	3	x	x	x		x			x		
Invertebrates														
Epifauna (trawl)	15	5	3	3	x	x		x		x				
Epifauna (video)	14	5	3	3	x	x		x		x				
Infauna	13	5	1	3	x	x		x		x				
Pelagic	11	5	1	3	x	x		x						
Megafauna														
ESAS sampling (birds, sea mammals)	15	6	1	3	x	x		x						
Towed hydrophones	4	4	0	2	x	x		x						
Habitat description														
Towed/dropped camera	13	5	2	3						x				
Sidescan sonar	11	5	1	3						x				

	Scoring Attributes	MSFD Descriptor Related to Task Category												
Task Category	**Total Score**	**N**	**Low Score**	**High Score**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Multi beam echosounder	11	5	1	3						x				
Ground trothing	8	4	1	3						x				
Pollution														
Sinking litter	11	4												x
Floating litter	10	4												x
Pollution in the water column	10	5										x	X	x
Pollution in organisms	10	5										x	X	
Pollution in the sediment	9	5										x	X	x
Environmental conditions														
Weather conditions	6	4	0	3							x			
Sea state	6	4	0	3							x			

[1] **Descriptors: 1 Biodiversity; 2 Non-indigenous Species; 3 Commercial Fish; 4 Foodwebs; 5 Eutrophication; 6 Sea-floor Integrity; 7 Hydrographical Conditions; 8 Contaminants; 9 Food Safety; 10 Litter.**

6.5.1 Data collection for fish

WGECO felt that the highest data collection priorities for fish were for stomach analyses and the collection of additional biological data. These data types were seen to be important for assessing GES under MSFD descriptors 1, 2, 3, 4 and 8. One member gave these tasks a score of 0 on the basis of direct relevancy to practical applications within the MSFD. It was noted that data gathered from fish stomachs should be collected and processed comprehensively but that it may not be necessary to conduct annual assessments. Further discussion of the rationale and required accuracy is provided below for some tasks.

6.5.1.1 Rationale and required accuracy for stomach sampling

Descriptor 4 of the MSFD addresses the foodwebs. Currently, the only viable method to identify the ~~structure~~ key energy flows within the marine foodweb is stomach sampling (noting that features of the foodweb other than energy flow are encompassed by the Descriptor). There are alternatives, such as stable isotope analysis or the tracking of abundances, but these resolve the energy flows at a much coarser level. Stomach sampling campaigns (Year of the Stomach) were carried out by ICES in the early 1980s and 1990s. Comparisons of the two surveys show that the energy flows within the foodweb changes over time, and that we must expect the current energy flows to be very different from these historical records.

An important descriptor of marine foodwebs is the mean dietary diversity of fish and other organisms. Rossberg *et al*. (2010) proposed an index for mean dietary diversity and demonstrated for marine communities of fish and squid across the oceans, including historical data for the North Sea, that the value of this index depends little on the habitat investigated. Thus, reference levels for mean dietary diversity can be set without the need to rely on historical data. Unpublished theoretical arguments and simulation results suggest that the index responds to an imbalance of species richness between the trophic levels of prey and predators. Due to the lack of data, the current values of this index for European waters and their trends are not known. A detailed analysis of the sampling effort required to determine the value of the mean dietary diversity index to a given accuracy is available (Rossberg *et al*., 2006).

An immediate need to determine the foodweb including fish by stomach sampling arises from Descriptor 3 (commercially exploited fish). The descriptor specifies that fishing mortalities should not exceed the levels required to attain Maximum Sustainable Yield (MSY). Strong trophic interactions among fish, as well as interactions resulting from competition for shared resources at lower trophic levels imply that MSY in a multispecies setting cannot be determined without knowledge of the foodweb. A detailed analysis of the sampling effort required to attain a given level of accuracy for multispecies modelling has been carried out by ICES Study Group on Multispecies models of the North Sea (SGMSNS 2006). Even when it would be impractical to sample foodwebs to the accuracy required for reliable long-term model predictions, which a straight-forward calculation of multispecies MSY requires, any available data on feeding preferences would provide important orientation for reaching MSY targets using more heuristic, adaptive management approaches.

Stomach data are also required for parameterization of Ecopath with Ecosim (EwE) models (Christensen *et al*., 2005), which have been used for investigating community dynamics under different scenarios. Up-to-date stomach data could allow for parameterization of EwE models that give more accurate predictions.

6.5.1.2 Rationale and required accuracy for individual condition

Descriptor 4 (foodwebs) specifies a characteristic "Productivity (production per unit biomass) of key species or trophic groups". For some species (e.g. birds) production can be measured by assessing the yearly number of offspring. For other species, such as fish, somatic growth can be just as important for production as population growth. Somatic growth has to be determined indirectly. There are indications that "fat", well-fed individuals are more productive than "slim" individuals, with respect to both somatic and population growth. The physical condition of fish, e.g. in terms of the condition factor body mass/length^3, can therefore serve as a measure for productivity. In community simulations conducted at Queen's University Belfast (T. Fung), productivity showed a coherent response to fishing pressure. It is unclear if sufficient data are available to determine baselines for condition factors for specific stocks or species. Such data would also allow determination of the accuracy at which condition would need to be measured to serve as an index for trophic flows.

6.5.1.3 Rationale for disease/parasite registration

Parasites are common in ecological networks, and many have complex life cycles and lower resource diversity than consumers (Dobson *et al*., 2009). Thus, they could promote secondary extinctions or regulate abundances (Dobson *et al*., 2009). However, they are inadequately represented in ecological networks that have been studied (Dobson *et al*., 2009). Registration of parasites would help provide data to include them in foodweb models that explore how parasitism affects foodweb properties (an example of an aquatic model with parasites is Arias-Gonzalez and Morand, 2006).

6.5.1.4 Rationale for lipid content

Lipid content data can be used for parameterization of the size-structured model by Hartvig *et al*. (2011). One of the parameters for this model is the fraction of the body weight containing energy reserves, which is a key determinant of the starvation mortality rate used in the model. Lipid content data can be used to provide more accurate estimates for model use.

6.5.2 Data Collection for Physical and Chemical Oceanography [CTD, chlorophyll, oxygen, nutrients, turbidity, etc.]

In relation to the physical and chemical oceanographic task category, WGECO gave highest priority to continuous measurements taken during the course of the survey to correlate with the biological data. Data on nutrients were also given a high total score for its value in linking physical conditions to primary productivity and foodwebs. Data from fixed stations was given a high rank due to the added value of time-series data in detecting long-term trends. Autonomous vehicles were seen as important and cost-efficient data collection devices of particular value when surveys have multiple tasks to achieve in a limited number of days at sea, or when data (usually bathymetry and temperature) are needed for areas covered by ice or otherwise inaccessible to the survey platform.

6.5.3 Data Collection for Biological Oceanography

Community models and theory indicate that there are strong quantitative and reproducible linkages between species richness at adjacent trophic levels. Knowledge of biodiversity at lower trophic levels should then help in interpreting biodiversity among fish. Resolution at species level is required. A well-defined sampling protocol should be followed so that species richness estimates derived from the sample are comparable.

6.6 Lessons learned from progress so far. Best practice guidance

6.6.1 OSPAR and ICES

The concept of Good Environmental Status (GES) is at the core of the MSFD. This needs to be achieved for eleven descriptors. In order to determine whether GES is achieved for each of these descriptors a process was initiated aimed at establishing the main attributes of these descriptors and selecting one or more potential indicators for these descriptors and their attributes.

An indicator can be considered a specific characteristic of a GES criterion (such as, for example, indicator 1.5.1 habitat area which is one of two listed indicators for the criterion habitat extent) that can either be qualitatively described or quantitatively assessed to determine (i) whether that criterion meets good environmental status or not or (ii) to ascertain how far that criterion departs from GES (OSPAR ICG-COBAM draft).

Indicators can therefore be used for two purposes within the framework of the Directive. Firstly, to assess environmental condition (state) and the extent to which good environmental status is being achieved with respect to any particular GES criterion (Article 9). Secondly, for the purposes of Article 10, to reflect achievement of environmental targets. Some indicators may serve both purposes at the same time.

The next phase of the implementation of the MSFD requires the setting of target values for indicators. The MSFD refers to such values as 'environmental targets`, but they are also identified under a variety of other names. In Annex I of the MSFD (referred to in Articles 3(5), 9(1), 9(3) and 24), the values are described as *levels* or *limits*. Annex IV (referred to in Articles 10(1) and 24) states that *reference points* (*target* and *limit reference points*) should be taken into account when setting environmental targets where appropriate (Cardoso *et al*., 2010).

'Environmental target', according to Article 3, means "a qualitative or quantitative statement on the desired condition of the different components of, and pressures and impacts on, marine waters in respect of each marine region or subregion". Article 10 requires that "Member States shall, in respect of each marine region or subregion, establish a comprehensive set of environmental targets and associated indicators for their marine waters so as to guide progress towards achieving good environmental status in the marine environment, taking into account the indicative lists of pressures and impacts set out in Table 2 of Annex III, and of characteristics set out in Annex IV". Environmental targets are hence specific requirements to be fulfilled on the way to achieving the overall aim, GES (OSPAR ICG-COBAM draft).

There is a difference between targets and reference points as used in fish stock management. Targets are human constructs, often resulting from political process expressing societal values. The concept of limit reference levels (or points) corresponds to features that are intrinsic to the ecosystem and hence are not human constructs but the results of natural processes.

In the OSPAR context "baseline" is probably the equivalent to reference point or level as it is used in the fisheries management context, i.e. "a specific value of state, against which subsequent values of state are compared: essentially a standard (articulated in terms of both quality and/or quantity) against which various ecological parameters can be measured" and against which the target can be set (OSPAR ICG-COBAM draft). However, where the fisheries management points as used by ICES are about keeping stocks *'..within safe biological limits..', i.e.* (1) exploited sustainably consistent with high long-term yields and (2) have full reproductive capacity or achieving a sustainable exploitation, within the OSPAR context "baseline" is often used interchangeably with "reference condition´ as used in the Water Framework Directive, i.e. a state at which the anthropogenic influences are negligible for all species or habitats (see Figure 6.6.1). Thus, there can be several baselines or reference levels against which the target is set. Considering that according to the MSFD, GES implies a state of the marine environment that corresponds to sustainable exploitation, sustainability-based reference levels are probably more useful that those based on pristine conditions.

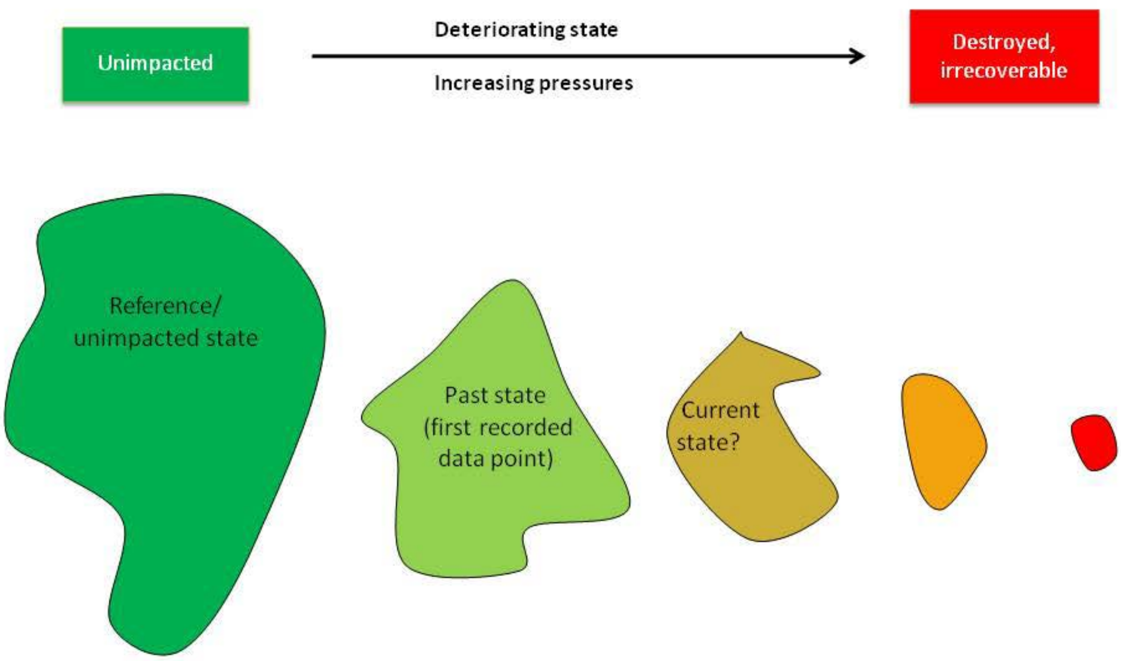

Figure 6.6.1. Reference/unimpacted state, past state (which could represent a sustainable level) along a gradient of deterioration (OSPAR ICG-COBAM draft).

The OSPAR ICG-COBAM draft distinguishes three different approaches to setting baselines:

- <u>Method A (unimpacted state/negligible impacts)</u> - Baselines can be set as a state at which the anthropogenic influences on species and habitats are considered to be negligible. This state is also known as 'reference conditions'. In this section, in order to be concise, this is referred to as 'reference/unimpacted state'.
- <u>Method B (past state)</u> - Baselines can be set as a state in the past, usually the point at which data collection on a specific species or habitat began.
- <u>Method C (current state)</u> - The date of introduction of an environmental directive or policy can be used as the baseline state, typically expressed as no deterioration from this state.

In the document these approaches are described in more detail. For method A three methods of determining this unimpacted state are distinguished: (1) based on existing reference conditions, (2) based on historical reference conditions or (3) modelling. We consider these approaches sensible but remark that the way it is presented here suggests as if the three methods mentioned under Method A can only be used to determine the unimpacted state while these methods can just as well be used to determine a sustainably exploited state. In fact any baseline for which there is a clear and unambiguous understanding of what it entails can be used as a baseline that helps set a target.

Once an appropriate baseline has been established, environmental targets (for both pressure and state) can then be generated according to the following 'target-setting options' (OSPAR ICG-COBAM draft):

- Method 1. Directional or trend-based targets
  - direction and rate of change
  - direction of change only
- Method 2. Targets set as an absolute Value
  - target set as baseline
  - target not set at baseline
- Method 3. Target set as a deviation from baseline

Again, these methods appear sensible and comprehensive and for further detail we refer to the original document.

Thus, once an indicator or suite of indicators has been selected the next particularly important step to implementation of the MSFD Descriptors is to establish the position on an indicator at or beyond which "good environmental status" has been achieved: i.e. the target level. Piha and Zamboukas (2010), reviewing the Methodological Standards Related to the Marine Strategy Framework interpret an environmental target as a value set on the basis of an environment indicator or index at or beyond which good environmental status has been achieved, or which guides progress towards achieving GES. This target is set relative to some reference level or baseline and is essentially a societal decision not necessarily based on ecological considerations only (Cardoso *et al*., 2010). Considering that according to the MSFD " *Member States should cooperate to ensure the coordinated development of marine strategies for each marine region or subregion*" this implies that this target level should be set for the (sub)region in close collaboration with the member states sharing that marine (sub)region.

Management must try to achieve at least that target level in order to qualify as GES. Under the MSFD it is necessary that these targets levels for delineating GES (or the reference levels they are based on) reflect ecologically comparable states. However, that does not require the same value everywhere; rather the target level should be scaled to local conditions while maintaining a consistent ecological meaning. In addition, society may set targets that are more ambitious than the ecologically determined reference levels, to fulfil their values and aspirations.

6.6.2 Member States

The MSFD implementation process in Germany and the UK involves multiple expert groups at different hierarchical levels ranging from the steering to a working level. Members of the expert groups comprise numerous agencies and institutions representing government and German federal states with different expertise. Currently, one of the main challenges for the expert groups is for instance the translation of the alignment of the concepts such as favourable conservation status and sustainable use into practical methods. This includes also a structured concept for the GES assessment. Based on the work by MS and Borja *et al*. (2011) Table 6.6.2.1 provides a summary of the work towards the initial assessment of the MSFD with a column showing commonalities and views of WGECO.

Issues highlighted in the comparison between national approaches also include the way in which interpretation of GES by one Member State can remain compatible with that of another neighbouring nation. For example, work to integrate a suite of biodiversity state indicators into a single measure that responds to management action is laudable. However the process for ensuring that this is consistent with other approaches in the same subregion is unclear, and Regional Seas Conventions should support this activity.

Table 6.6.2.1. Summary of the work towards the initial assessment of the MSFD done by two member states (UK and Germany) and a study in the Bay of Biscay by Borja *et al.* (2011). The work of the two member states is still ongoing. The WGECO column highlights commonalities and differences between these approaches.

Topic	UK	Germany	Borja *et al.* (2011)	WGECO comment
Monitoring needs	Starting point is existing monitoring programmes to fulfil requirements (e.g. WFD, HD, CFP).	Starting point is existing monitoring programmes to fulfil requirements (e.g. WFD, HD, CFP).	Starting point is existing monitoring programmes to fulfil requirements (e.g. WFD, HD, CFP).	Always based on existing monitoring programmes to fulfil requirements (e.g. WFD, HD, CFP).
Development of indicators	Distinguish different components of the marine ecosystem: seabirds, marine mammals and turtles, sediment habitats, rock and biogenic reef habitats, fish and cephalopods, and pelagic habitats. Provide for each component suitable indicators for the four biodiversity descriptors. Depending on monitoring programmes determine which indicators per descriptor (e.g. distributional range) can be calculated and combine with most appropriate species.	Relevant contents of the long-term monitoring programmes are related to (some of) the indicators of each of the descriptors. Grouping based on four scales of ecosystem component (species, functional groups, habitat types, ecosystems), which may be subdivided into a number of key elements (e.g. individual species).	Grouping marine ecosystem components into four distinct (but interlinked) systems: (i) water and sediment physico-chemical quality (including general conditions and contaminants); (ii) planktonic (phyto- and zooplankton); (iii) mobile species (fishes, sea mammals, seabirds, etc.) and (iv) benthic species and habitats	Similar approaches (i.e. based on known monitoring programmes, MSFD indicators coming from the task groups and some grouping of ecosystem components and/or attributes that are combined with the MSFD indicators. Only the groups chosen differ.
Establishing thresholds and reference levels	scientific understanding underlying some indices may be insufficient for setting targets even when good baseline data are available	alignment with assessment methodologies in WFD, HD, HELCOM	Methods used to identify thresholds and reference levels are varied and depend on the characteristics of each descriptor. Some weighted integration/aggregation is applied.	No best practice has emerged. Some commonality in first approaches. Insufficient information to evaluate the methods applied by Borja. Yet to be further developed.
Management measures	Currently gathering information on potential management measures. Evaluation according to effectiveness in alleviating pressures or effectiveness in helping to achieve GES. Choice based on Cost–benefit Analysis	To be done. Definition of targets and GES will be linked to manageable human activities	Not considered. But assessment of the environmental status allows the development of management plans for reducing the human pressures	Thus far only considered by UK.

6.6.3 WGECO considerations

For a number of criteria and related indicators, EC (2010) indentifies "the need for further development and additional information [...] to be further addressed in the process for the revision of this Decision." To stimulate such developments WGECO notes here to a number of conceptual distinctions which are not yet fully developed in the Directive. Some distinctions were highlighted already in Section ["OSPAR and ICES"]. Below we point to the need to differentiate with respect to the purpose of indicators, conceptions of biodiversity, and notions of foodwebs.

Indicators can serve policy, management, and surveillance purposes (WGECO 2004; Heink and Kowarik, 2010). This distinction is here illustrated for Descriptor 3 (commercially exploited stocks), where it is well understood; but similar differentiations could be necessary for other Descriptors. Most fisheries are currently managed at the level of individual stocks. Indicators relevant to a PSR scheme are, e.g. standing stock biomass (SSB) and fishing mortality. To understand interactions between SSB and fishing mortality it is important to monitor also details of stock structure. For management purposes, there is no immediate need to integrate these indicators to higher levels. But it can have high policy relevance, as demonstrated by the frequently cited statistic by FAO comparing the numbers of underexploited, fully exploited, and collapsed stocks. Criteria for the policy relevance of indicators (e.g. relevance to the public, simplicity, the total number of indicators) are not always aligned with the characteristics of good management indicators (e.g. predictability or responsiveness to management measures). This may lead to different strategies when making indicators operational.

There is a widely acknowledged differentiation between the public conception of biodiversity (emphasizing charismatic megafauna) and the scientific ideas (applied much broader). The Marine Strategy serves both policy and management purposes, so that both aspects should be taken into account. Confounding the two, however, might not serve either purpose well.

EC (2008) acknowledges the importance of marine foodwebs by inclusion of Descriptor 4. In addressing this Descriptor, awareness of two different conceptions of foodwebs that have emerged in the scientific literature could be helpful. The first conception emphasizes energy flows and interactions between functional groups (e.g. Werner *et al.*, 2007). The second conception emphasizes the sparse and highly complex network of feeding interactions between individual species (Cohen *et al.*, 1990). While these two are related, the characteristic structural and dynamical features are very different. Management of the "functional" foodweb is more important in the short term. The balance of nature in the "species" foodweb is more delicate (as demonstrated by high parameter sensitivity of models, see Yodzis, 1998), it is closely linked to biodiversity (May, 1972), and the characteristic time-scales tend to be longer (Rossberg and Farnsworth, 2010). Both aspects might need to be covered when addressing Descriptor 4.

6.7 References

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Annex 1: ToR d-Sections in WGECO reports relevant to move the MSFD forward

WGECO 2005

Chapter 3 Science Advice to Support the European Marine Strategy

3.1 The Policy and ICES context

3.2 Integrated Management – Science needs and implications

3.3 Regional Management – Science needs and implications

3.4 Additional New challenges for science and advice in support of the European Marine Strategy

- What constitutes a Healthy Ecosystem
- How should Conservation Limits be identified?
- Scenario Modelling and Risk Management
- Selecting Appropriate Suites of objectives
- Developing Management Strategies in Integrated Management Framework

Chapter 6 Indicators and their application in a management framework

6.2 Selection of ecosystem indicators

- State indicators
- Pressure indicators
- Potential indicators and their evaluation
- Evaluation of indicators
- The process of indicator selection

6.3 Application of indicators in the new ICES Advisory framework

- Indicators and Objectives
- Indicators and Reference points
- Function of Indicators in the New ICES Advisory Framework

6.4 Application of indicators in a management framework

6.5 Fish and invertebrate taxa which are appropriate to use as indicators of habitat quality

- What constitutes a habitat
- What is habitat quality
- WGECO's consideration of the scientific aspects of marine benthic habitat quality

6.6 The approach and methods

- Criteria
- The selection of potential indicator taxa

WGECO 2006

Chapter 3 The effects of fishing on the North Sea ecosystem

3.2 Ecosystem components identified by WGECO

3.7 Community level metrics of the effects of fishing on ecosystem properties

Chapter 4 Assessing the key pressures on marine ecosystems

4.2 Key pressures and ecosystem components

- Defining the approach
- Ecosystem components
- Pressures
- Weighting the significance of interactions between pressures and ecosystem components
- what are the key pressures?
- Metrics, Indicators, Dataseries and Reference levels for Key Pressures

4.3 Defining the uses of the Integrated Assessment framework

- Using indicators within these approaches
- Using indicators of key pressures to investigate change in state
- Linking change in state to key pressures (steps (i) and (ii))
- Identifying activities contributing to key pressures (step (iii))
- Comment on preparedness to undertake comprehensive assessments of ecosystem state in the North Sea

Chapter 6 Development of EcoQO on changes in the proportion of large fish and evaluation of size-based indicators

- This has been an ongoing work of WGECO and has led to the use of the LFI in the DCF and MSFD

WGECO 2007

Chapter 2 Applicability of the "3-stage Model"

- The 3-stage model" was designed originally for fishery harvest control rules and evaluated in the context of ecosystem indicators

Chapter 4 Size-based EcoQOs for Fish Communities – Continuation of LFI work

Chapter 7 Assessment of the environmental impact of marine fisheries

- Actions and measures
- Driving forces, pressures and impacts
- State of the marine environment
- Impacts from fishing

Chapter 8 Changes in biota caused by hydrographic and sea temperature changes

Based on inputs from various ICES ecosystem WG, and provided as evidence of the need for surveillance style indicators to chart the changes in background conditions necessary for the setting of indicators and thresholds within the MSFD

- Phytoplankton and zooplankton communities
- Angiosperms and macroalgae
- Benthic invertebrates
- Fish communities
- Marine mammals, reptiles and seabirds

Perspectives on understanding changes in biota in response to environmental change

- Direct or indirect effects,
- Demonstrating causality
- Regional effects
- Climate variation vs. climate change

WGECO 2008

WGECO was heavily involved in developing the approaches to be used in the OSPAR 2010 Quality Status Report, particularly in terms of the activity/pressure relationships evaluation matrix, and thresholds between status categories. The last two were subsequently systemised in the Robinson *et al*. approach.

Chapter 3 Changes in the distribution and abundance of marine species in relation to climate change for the 2010 OSPAR QSR

- Oceanographic background
- Detection of effects of climate change on marine biota – linked to Chapter 8, WGECO 2007
- Groups covered:
  - Plankton
  - Benthos
  - Fish
  - Seabirds
  - Marine mammals
  - Invasive species

Chapter 4 Draft environmental impact of marine fisheries for the 2010 OSPAR QSR

- The development of fisheries management and policy since 1998
- Fishing activities in the OSPAR maritime area
- Impacts of fisheries on the ecosystem
- Assessment of fisheries measures and their effectiveness

4.2 Regional QSR: Arctic, Greater North Sea, Celtic Seas, Bay of Biscay and Iberia, and Wider Atlantic

Chapter 7 Assessment matrix of pressure of human activities and ecosystem components

The Robinson *et al*. approach for OSPAR QSR 2010

WGECO 2009

Chapter 3 DG MARE Special Request; Fisheries Indicators

3.2 Background to the fisheries indicators – these are indicators of the extent and level of fishing activity for collection under the DCF and anticipating use in the MSFD

3.3 Approach to data collection: Logbook data, VMS data, Analytical methods

3.5 Data needs for determining fishery indicators: VMS data, Logbook data for >15 m vessels, Landings data

Chapter 4 Bringing consistency in the use of ecological terms and concepts in marine ecosystem management

It was recognized that there was a proliferation of terminology in the context of marine ecology and marine ecosystem management, and that it would be useful to bring some order to this, and to provide consistent interpretations that would be useful *inter alia* for the operation of the MSFD. The aim of this piece of work then was to provide a consistent set of ecosystem terminology and identify synonyms and overlaps, in the context of, existing agreements, and also undefined concepts:

- Ecological concepts
- Terms with multiple meanings
- Ecosystem descriptors/status
- Environmental management strategies
- Human impacts/pressures

Chapter 6. Applying risk-based methodologies to assess degree of impact

The Robinson *et al*. (REA) methodology, and review of the Utrecht OSPAR QSR Chapter 11 Workshop.

Chapter 8 Prioritizing fish species for research on fishing mortality

Fishing mortality is one of the key pressures on the ecosystem and is relevant to MSFD indicators 1, 3, and 4. Currently, F is estimated mainly for the key commercial species, and this analysis aimed to identify species for which a quantification of F would important for MSFD and wider ecosystem based fisheries management. On this basis the estimation of fishing mortality was considered important for:

- High biomass species
- Vulnerable species
  - Species that are vulnerable because of their life-history characteristics or ecology
  - Species that are vulnerable as a consequence of high catchability
- Commercially important non-assessed species

- Species that have exhibited unexplained population declines
- Species listed as being of concern by conservation agencies

WGECO 2010

Chapter 3 Integrated ecosystem assessments

3.3 Types of assessment(s) needed for implementing the MSFD

3.4 Integrated ecosystem assessments: existing frameworks:

- OSPAR approach
- REGNS approach
- United States approach
- Canadian approach

3.5 The way forward - the initial assessment:

- Evaluation of ecosystem components
- Evaluation of the pressures
- Use of a framework to identify key pressures and components
- Selection of indicators

WGECO has evaluated Integrated Ecosystem Assessments on a number of occasions.

Chapter 4 Data analyses required to examine the relationships between perturbation and recovery capacity

4.1 Recovery and resilience:

- Recovery used for populations
- Recovery used above the population level
- Resilience

4.2 Recovery in the context of the MSFD

4.3 Analyses required to examine the relationships between perturbation and recovery

Chapter 6 Review of methods used to determine "good environmental status"

6.2 Comparison of Water Framework Directive, Habitat Directive, and Marine strategy Framework Directive

6.3 Moving forward with the MSFD

- How to choose suites of indicators from the large candidate set
- How to set reference levels on the chosen indicators
- How to combine information across indicators for an overall assessment of "good environmental status"

6.4 Processes for the next step

- Considerations from assessment of assessments
- Assessment process issues and the MSFD

Chapter 8 Extending marine assessment and monitoring framework used in Chapter 10 of the QSR 2010 (OSPAR request 2010/1)

8.2 Improvements to the thresholds between different assessment classes

- Overview of OSPAR QSR approach
- Scientifically robust thresholds between different assessment classes (example for the fish community)
- Scientifically robust thresholds between different assessment classes (other components)
- Former natural conditions-constraint on reasonable use of data
- WGECO approaches to defining thresholds

8.3 Extending the methodology to support the assessment of plankton communities;

8.4 Improving the method for working at different scales,

- E.g. OSPAR Region, and subregions such as the Irish Sea or the Channel or the level of an estuary or an MPA.

Chapter 9 Environmental interactions of wave and tidal energy generation devices

9.2 Direct effects:

- Habitat change
- Water column processes and hydrography
- Exclusion zones
- Noise
- Electromagnetic fields (EMFs)
- Contaminants and anti-fouling

9.3 Indirect effects:

- Food chain
- Reproduction and recruitment

9.4 Principle areas of environmental risk and the scope for mitigation

Annex 2: Details as specified in the HD and WFD that provide methodological standards for setting reference levels in the MSFD

Habitats Directive (92/43/EEC)

Table 1. Methodological standards related to the Habitat Directive that are available for the GES definitions according to different descriptors and indicators (Piha and Zampoukas, 2011; modified).

Descriptor	Indicator		Source	Remarks
1. Biological Diversity	Species distributional range	1.1.1	(EC, 2006)	Not available for all species and habitats
	Distributional range of habitat	1.4.1		Not available for all species and habitats
	Habitat area	1.5.1		Not available for all species and habitats
	Area covered by the species (for sessile/benthic species)	1.1.3		Not available for all species
	Population abundance and/or biomass, as appropriate	1.2.1.		Not available for all species
	Distributional pattern of habitat	1.4.2		Only some recommendations
	Condition of the typical species and communities of the habitat	1.6.1		Only some rough guidelines
6. Sea Floor	Type, abundance, biomass and areal extent of relevant biogenic substrata	6.1.1		Applicability for biogenic substrata must be evaluated

The concept of ‘favourable conservation status’ (FCS) constitutes the overall objective to be reached for all habitat types (Art. 1.e, 92/43/EEC) and species (Art. 1.i, 92/43/EEC) of community interest. FCS can be described as a situation where a habitat type or species is prospering (in both quality and extent/population) and with good future potential.

FCS is assessed across all national territory (or by biogeographical region within a country where two or more regions are present) and should consider the habitat or species both within the Natura 2000 network and in the wider countryside.

For habitat types and species typically from the marine environment, Member States should report about their conservation status using the following marine regions:

- Atlantic: Northern and Western Atlantic, from the Straits of Gibraltar to the Kattegat, including the North Sea;
- Baltic: east of the Kattegat, including the Gulf of Finland and the Gulf of Bothnia; Mediterranean: east of the Straits of Gibraltar;
- Macaronesian: Economic Exclusive Zones of the Azores, Madeira and Canary Archipelagos.

For the purpose of evaluating the Conservation Status of habitat and species each Member State needs to establish the following "favourable reference values" (FRV):

- the appropriate reference range and area for the habitats of Annex I;
- the appropriate reference range and population for the species of Annexes II, IV and V.

These reference values represent the baselines to which compare the present and projected Conservational Status of habitat/species.

According to the HD guidelines (EC, 2006), for all four Favourable Reference Values it is possible to carry out the assessment of Conservation Status by setting the FRV as 'greater than present day value' and this is preferable to using 'unknown' in cases where it is clear that the present day range, area or population is not sufficient but where it is not possible to estimate what the correct value should be.

HD guidelines (EC, 2006) also specify that for non-coastal marine species it is probably more sensible to set FRV for the whole marine region by the concerned Member States.

Favourable Reference Status

The following definitions of Favourable Reference Status for range (species and habitats), population (species) and area (habitat type) are given in the Habitat Directive guidelines (EC, 2006).

Favourable reference range

Range within which all significant ecological variations of the habitat/species are included for a given biogeographical region and which is sufficiently large to allow the long-term survival of the habitat/species; favourable reference value must be at least the range (in size and configuration) when the Directive came into force; if the range was insufficient to support a favourable status the reference for favourable range should take account of that and should be larger (in such a case information on historic distribution may be found useful when defining the favourable reference range); 'best expert judgement' may be used to define it in absence of other data.

The background information and parameters may be useful to set Favourable Reference Range (FRR) for both species and habitats include:

- Current range;
- Potential extent of range taking into account physical and ecological conditions (such as climate, geology, soil, altitude);
- Historical range and causes of change;
- Area required for viability of habitat/species, including consideration of connectivity and migration issues.

Favourable reference population

Population in a given biogeographical region considered the minimum necessary to ensure the long-term viability of the species; favourable reference value must be at least the size of the population when the Directive came into force; information on historical distribution/population may be found useful when defining the favourable reference population; 'best expert judgement' may be used to define it in absence of other data.

The background information and parameters that may be useful to set FRP are:

- Historical distribution and abundances and causes of change;
- Potential range;
- Biological and ecological conditions;
- Migration routes and dispersal ways;
- Gene flow or genetic variation including clines;
- Population should be sufficiently large to accommodate natural fluctuations and allow a healthy population structure.

Favourable reference area

Total surface area in a given biogeographical region considered the minimum necessary to ensure the long-term viability of the habitat type; this should include necessary areas for restoration or development for those habitat types for which the present coverage is not sufficient to ensure long-term viability; favourable reference value must be at least the surface area when the Directive came into force; information on historical distribution may be found useful when defining the favourable reference area; 'best expert judgement' may be used to define it in absence of other data.

The background information and parameters that may be useful to set FRA:

- Historical distribution and causes of change;
- Potential natural vegetation;
- Actual distribution and actual variation;
- Dynamics of the habitat type;
- Natural variation should be fully covered (subtypes, syntaxa, ecological variants, etc.);
- Distribution pattern should allow exchange/gene flow in typical species.

Conservation Status Evaluation: targets and limits

The Appendix 1 of the guidelines and explanatory notes on the Art. 17 of the Habitat Directive (EC, 2006) defines a quantitative methodology for the assessment of the Conservation Status of a species (Annex C) and habitats type (Annex E) based on the following parameters:

- Species (per biogeographic region within a MS): range, population, habitat for the species, future prospects;
- Habitat Type (per biogeographic region within a MS): range, area covered by habitat type within range, specific structures and functions (including typical species); future prospects.

Three classes of Conservation Status are used. 'Good': where the species or habitat is at FCS as defined in the Directive and the habitat or species can be expected to prosper without any change to existing management or policies. Two classes of 'Unfavourable' are recognized: one 'Unfavourable-Bad' (red) where the habitat or species is in serious danger of becoming extinct (at least locally) and 'Unfavourable-Inadequate' (amber) for situations where a change in management or policy is required but the danger of extinction is not so high.

It is worth noting that threshold between "Unfavourable" and "Unfavourable-bad" status is based on a quantitative assessment. Quantitative thresholds are related to temporal trends (e.g. population range large decline: loss of more than 1% per year within period specified by MS) or percentage change (e.g. population range large decline more than 10% below the favourable reference point) of present date parameters compared with the favourable reference conditions.

Table 2. General evaluation matrix (per biogeographic region within MS for assessing conservation status of species (EC, 2006; Appendix I, Annex C; modified).

Parameter	Conservation Status			
	Favourable ('green')	Unfavourable - Inadequate ('amber')	Unfavourable - Bad ('red')	Unknown
Range	Stable (loss and expansion in balance) or increasing AND not smaller than the 'favourable reference range'	Any other combination	Large decline: Equivalent to a loss of more than 1% per year within period specified by MS OR more than 10% below favourable reference range	No or insufficient reliable information available
Population	Population(s) above 'favourable reference population' AND reproduction, mortality and age structure not deviating from normal (if data available)	Any other combination	Large decline: Equivalent to a loss of more than 1% per year (indicative value MS may deviate from if duly justified) within period specified by MS AND below 'favourable reference population' OR More than 25% below favourable reference population OR Reproduction, mortality and age structure strongly deviating from normal (if data available)	No or insufficient reliable information available
Habitat for the species	Area of habitat is sufficiently large (and stable or increasing) AND habitat quality is suitable for the long-term survival of the species	Any other combination	Area of habitat is clearly not sufficiently large to ensure the long-term survival of the species OR Habitat quality is bad, clearly not allowing long-term survival of the species	No or insufficient reliable information available
Future prospects (as regards to population, range and habitat availability)	Main pressures and threats to the species not significant; species will remain viable on the long term	Any other combination	Severe influence of pressures and threats to the species; very bad prospects for its future, long-term viability at risk.	No or insufficient reliable information available
Overall assessment of CS	All 'green' OR three 'green' and one 'unknown'	One or more 'amber' but no 'red'	One or more 'red'	Two or more 'unknown' combined with green or all "unknown"

Table 3. General evaluation matrix (per biogeographic region within MS) for assessing conservation status of habitat type (EC, 2006; Appendix I, Annex E. modified).

Parameter	Conservation Status			
	Favourable ('green')	Unfavourable - Inadequate ('amber')	Unfavourable – Bad ('red')	Unknown
Range	Stable (loss and expansion in balance) or increasing AND not smaller than the 'favourable reference range'	Any other combination	Large decrease: Equivalent to a loss of more than 1% per year within period specified by MS OR More than 10% below 'favourable reference range'	No or insufficient reliable information available
Area covered by habitat type within range	Stable (loss and expansion in balance) or increasing AND not smaller than the 'favourable reference area' AND without significant changes in distribution pattern within range (if data available)	Any other combination	Large decrease in surface area: Equivalent to a loss of more than 1% per year (indicative value MS may deviate from if duly justified) within period specified by MS OR With major losses in distribution pattern within range OR More than 10% below 'favourable reference area'	No or insufficient reliable information available
Specific structures and functions (including typical species)	Structures and functions (including typical species) in good condition and no significant deteriorations / pressures.	Any other combination	More than 25% of the area is unfavourable as regards its specific structures and functions (including typical species)8	No or insufficient reliable information available
Future prospects (as regards range, area covered and specific structures and functions)	The habitats prospects for its future are excellent /good, no significant impact from threats expected; long-term viability assured.	Any other combination	The habitats prospects are bad, severe impact from threats expected; long-term viability not assured.	No or insufficient reliable information available
Overall assessment of CS	All 'green' OR three 'green' and one 'unknown'	One or more 'amber' but no 'red'	One or more 'red'	Two or more 'unknown' combined with green or all "unknown"

Water Framework Directive (2000/60/EC)

Table 4. Methodological standards sources related to Water Framework Directive (2000/60/EC) that are available for the GES definitions according to different descriptors and indicators (Piha and Zampoukas, 2011; modified).

Descriptor	Indicator		Source	Remarks
1. Biological Diversity	Area covered by the species (for sessile/benthic species)	1.1.3	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only Coastal
	Population abundance and/or biomass, as appropriate	1.2.1.	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only Coastal
	Relative abundance and/or biomass of the habitat, as appropriate	1.6.2	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only coastal/only for some species
	Composition and relative proportions of ecosystem components (habitats and species)	1.7.1.	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only coastal/only for some species
	Physical, hydrological and chemical conditions of the habitat	1.6.3	Eutropphication Guidance EC, 2009	Only for coastal waters. No methods for hydrological conditions
4. Foodwebs	Abundance trends of functionally important selected groups/species	4.3.1	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only coastal/only for some species
6. Sea Floor	Type, abundance, biomass and areal extent of relevant biogenic substrata	6.1.1	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only coastal
	Presence of particularly sensitive and/or tolerant species	6.2.1.	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only coastal
	Multimetric indices assessing benthic community condition and functionality, such as species diversity and richness, proportion of opportunistic to sensitive species	6.2.2	Birk *et al*., 2010; Carletti and Heiskanen, 2009	Only coastal

The WFD assessment methods for all quality elements are listed and described in the online database compiled within the WISER project (Birk *et al.*, 2010).

Regarding coastal waters, the database includes information on the assessment methods for phytoplankton, macrophytes, macroalgae, zoobenthos, including metrics of abundance and biomass, presence of sensitive and or/tolerant species.

However, their compliance with the WFD is not yet fully checked and their application away from coastal waters needs to be evaluated or/and developed for the purposes of GES. A description of some of the above mentioned methods were included in the Commission Decision 2008/915/EC and can be found in the technical report of the 1st intercalibration phase (Carletti and Heiskanen, 2009).

The WISER database (available at http://www.wiser.eu/programme-and-results/data-and-guidelines/method-database/) was accessed in order to provide a list of all methods already addressed for the evaluation of ecological status in coastal waters in different MS. The analysis shows that most of MS completed at least a first evaluation of the biological quality elements, a part from angiosperms where many MS still need to carry out an assessment (Table 1.2.2.7).

Details about the metrics and methodologies adopted (including boundary limits set to define the boundaries between different water quality status) can be found in the above mentioned WISER database.

Table 5. Assessment methods available for phytoplankton, angiosperms, macroalgae, zoobenthos for coastal waters in MS according to the WFD activities Blue cells shows those MS that submitted their assessment methods (Elaboration from WISER Database, accessed on April 16th, 2011).

Geographical Intercalibration Groups	Member State	Angiosperms	Macroalgae	Phytoplankton	Benthic Invertebrates
Northeastern Atlantic	Belgium				■
	Denmark	■	■	■	■
	France		■	■	■
	Germany	■	■	■	■
	Ireland	■	■	■	■
	Netherlands	■	■	■	■
	Norway		■		■
	Portugal		■		■
	Spain		■	■	■
	Sweden		■		■
	United Kingdom	■	■	■	■
Baltic Sea	Denmark	■	■	■	■
	Estonia	■		■	■
	Finland		■	■	■
	Germany	■	■	■	■
	Latria			■	■
	Lithuania		■	■	■
	Poland	■	■	■	
	Sweden	■	■	■	■
Mediterranean Sea	Cyprus		■	■	■
	France		■	■	
	Greece	■	■		■
	Italy	■	■	■	■
	Slovenia		■	■	■
	Spain	■	■	■	■
Black Sea	Romania		■	■	■

7 ToR E: Strategic Initiative on Biodiversity

e) ToR in relation to the Strategic Initiative on Biodiversity that is being developed by Simon Jennings and Mark Tasker.

Conservation of marine biodiversity and sustainable use of marine resources emerge as one major policy issue of the twenty-first century. An increasing amount of legislation and number of international conventions include marine biodiversity conservation among the highest ranked priorities. ICES has a long history and authority in providing marine biodiversity science and advice. However, this prominence is strongly linked with science and advice related to sustainable exploitation of fish populations. To enlarge the relevance of advice to the broader range required by the developing biodiversity policy, ICES is in a process to strengthen and better organize its activities in the realm of marine biodiversity. The ultimate purpose of this initiative is to develop and provide advice that makes trade-off between conservation and sustainable use visible and explicit.

To that effect, short- and medium-term priorities identified by the Workshop on Marine Biodiversity held in Copenhagen in February 2011 (ICES 2011) include the provision of guidance that promotes consistency and soundness of practices when evaluating environmental status (for the Marine Strategy Framework Directive). More specifically, WGECO is requested to provide guidance on defining reference points or reference conditions that correspond to sustainable use. WGECO 2010 provided guidance on methods to determine "good environmental status" and the development of integrated ecosystem assessment; this work is not repeated here. Rather this section examines how this general guidance applies to biodiversity. The trade-off between conservation and sustainable use might not be so explicit at a high level such as general objectives of international conventions or laws; however, it might appear when it comes to implementing policies with those objectives. A general framework for setting reference points in relation to sustainable use was developed by WGECO 2010. Here we outline the extent to which this framework is applicable, in particular, to biodiversity indicators.

This section goes in two steps:

- First WGECO comments on the potential trade-off between biodiversity objectives related to conservation *vs.* sustainable use.
- Second WGECO provides guidance to promote consistency and soundness of practices when evaluating environmental status, more specifically on defining reference points or conditions that correspond to sustainable use. The latter include methods to assess when components of biodiversity are subject to serious or irreversible harm, in order to guide the setting of limits for biodiversity indicators. Guidance is provided about consideration of changing conditions, and the potential role of experts in developing advice on ecosystem status and management.

7.1 Trade-off between conservation and sustainable use

7.1.1 Objectives of marine conservation and sustainable use of marine resources

This section summarizes the high-level objectives of conventions and acts that pertain to the conservation and sustainable use of marine habitats, fauna, and flora. The policy drivers related to marine biodiversity issues were taken directly from the WKMARBIO report (ICES 2011) supplemented with reference to the relevant acts and conventions.

Global conventions

The Convention on Biological Diversity (CBD) has three main objectives:

- conservation of biological diversity;
- sustainable use of biodiversity components;
- fair and equitable sharing of benefits arising from the use of genetic resources.

Subsequent processes under CBD elaborated goals for the marine environment, which are designed:

- to halt the loss of marine and coastal biological diversity nationally, regionally, and globally;
- to secure its capacity to provide goods and services.

A set of targets was agreed at the most recent (2010) meeting of the CBD (www.cbd.int); the most relevant of these to the conservation of marine biodiversity were:

- by 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, so that overfishing is avoided, and fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems.
- by 2020, at least 10 per cent of coastal and marine areas are conserved through ecologically representative and well connected systems of protected areas and other effective area-based conservation measures.

The United Nations Food and Agriculture Organization Code of Conduct for Responsible Fisheries (FAO 1995) advises fisheries management organizations to adopt measures that are designed to maintain or restore stocks at levels capable of producing maximum sustainable yield. Such measures should provide *inter alia* that:

i) excess fishing capacity is avoided and exploitation of the stocks remains economically viable;

ii) the economic conditions under which fishing industries operate promote responsible fisheries;

iii) biodiversity of aquatic habitats and ecosystems is conserved and endangered species are protected;

iv) depleted stocks are allowed to recover or, where appropriate, are actively restored;

v) adverse environmental impacts on the resources from human activities are assessed and, where appropriate, corrected.

The 1982 Convention Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Species (UN 1995reference missing) has as its overall objective to ensure the long-term conservation and sustainable use of straddling fish stocks and highly migratory fish stocks through effective implementation of the relevant provisions of the Convention.

Conventions in the ICES area

Both the Helcom and OSPAR Conventions have monitoring and assessment strategies that include biodiversity.

Article 1 of the Marine Strategy Framework Directive (MSFD) specifies, among other subjects, that "Marine strategies shall apply an eco-system based approach to the management of human activities,…while enabling the sustainable use of marine goods and services by present and future generations" (European Union 2008). The MSFD defines "Good Environmental Status" with a series of descriptors, several of which directly relate to the conservation and sustainable use of biodiversity (descriptors 1 to 6).

The Objectives of the Common Fisheries Policy (currently Regulation 2371/2002) are:

1) "To maintain fishing mortality at or below levels that are necessary to achieve maximum sustainable yield for all targeted stocks."
2) "To maintain or reduce fishing impact on the eco-system at or below sustainable levels."
3) "To develop a viable, economically efficient and globally competitive European fisheries and aquaculture industry."

The Common Fisheries Policy has multiple articles directly applicable to biodiversity conservation needs including "taking measures....to minimize the impact of fishing activities on marine eco-systems" (article 2). Priority areas for biodiversity conservation at present are:

- to reduce the overall fishing pressure to sustainable levels;
- protect sensitive marine habitats and sensitive species;
- avoid foodweb distortions;
- eliminate unwanted bycatches.

In the Northwest Atlantic, Canada passed the Oceans Act in 1997, with implementation supported by Canada's Oceans Strategy (2002). This strategy outlines how Canada's international commitments and domestic mandates for marine conservation would be met. The Department of Fisheries and Oceans has developed and implemented a Sustainable Fisheries Framework, intended to place all fisheries management in an ecosystem context, with specific policies for protection of special benthic habitats, for management of bycatches, and for fisheries on forage species.

The Magnuson-Stevens Fishery Conservation and Management Act, commonly referred to as the Magnuson-Stevens Act, is the primary law governing marine fisheries management in the USA. It includes provisions for protecting essential fish habitat as well as conserving and rebuilding fish stocks. Other US legislation related to biodiversity includes the Marine Mammal Protection Act and the Endangered Species Act.

7.1.2 Definitions

Many policy strategies have developed goals for biodiversity and sustainability comprising aspects of ecological, economical and societal wellbeing. Whether maintaining biodiversity and sustainable use are compatible first depends on the definition of these concepts. Many of these strategies provide their own definitions. WGECO 2009 collected and reviewed definitions for several policy terms, *inter alia* biodiversity and sustainability, from several high level sources (UN, FAO, OSPAR, EU, and HELCOM). We summarize this work below, with special emphasis on the terminology of the European Marine Strategy Framework Directive (MSFD), a pressing policy driver at the moment with the aim of achieving its goals by 2020.

Definitions

The goal for the MSFD Descriptor 1 (D1) is "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." A further mention of biological diversity appears in D4, which states that all ecosystem elements occur at normal abundance and diversity. This descriptor prescribes that species distribution ranges shall be within their bioclimatic envelopes, and abundances in line with their physiographic conditions. Full reproductive capacity and long-term persistence is indirectly linked to diversity but directly to levels of occurrence.

WGECO 2009 considered the CBD biodiversity definition ('The variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems') to be sufficiently comprehensive and robust to be used as a working definition.

We pick up definitions for *sustainability* from three documents, namely the CBD, MSFD and ICES 2005reference missing? In Article 2 of the Convention on Biological Diversity (CBD 1992) this term is defined as '*the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations*'. The ICES (2005) definition focused more on the resources by specifying that sustainable exploitation does not prejudice any future exploitation and it does not have a negative impact on the marine ecosystems. MSFD states that sustainable use means to safeguard the potential uses and activities by current and future generations and is an intrinsic element of the aim of 'good environmental status' (Art. 3 (5)).

Consistency among definitions and necessary links

The CBD definition of biodiversity is consistent with sustainable use. The focus is on variability, and as long variability is preserved, exploitation is implicitly permitted.

At the general level, the MSFD defines biodiversity in close relationship with sustainable use (MSFD Article 3, *Definitions*). From all attributes referenced in the MSFD definitions in D1 and D4, 'normal' has a strong normative power since it prescribes a certain level of population. The long-term persistence mentioned in D4 is less normative, since it only requires a stable population large enough to ensure full reproductive capacity. At least for fish stocks, full reproductive capacity does not preclude exploitation of the stock.

ICES (ICES 2005) related the term 'normal' to species population dynamics, considering that recovery after a disturbance should be expected to be 'rapid and secure'. 'Rapid' is applied taking account of the normal dynamics of the properties being monitored. ('Rapid' for herring is not the same as 'rapid' for beluga). WGECO 2009 agreed that the condition of 'rapid and secure' is an important aspect of recovery. WGECO 2010 builds on this agreement and links sustainable use with 'rapid and secure' recovery. A non-sustainable perturbation is defined as one from which recovery is not likely to be rapid and secure. This in turn relates to the element of ecosystem functioning in the MSFD, where systems must have the ability to function 'fully' and be able to maintain their resilience towards anthropogenic impacts.

7.1.3 To what extent are these objectives aligned? What is the trade-off between sustainable use and biodiversity conservation?

At a high level, the objectives of conventions governing the conservation and sustainable use of marine resources and their definitions appear to be closely aligned. The Convention on Biological Diversity recognizes the sustainable use of biodiversity components. In turn, the Code of Conduct for Responsible Fisheries advises measures to conserve biodiversity of aquatic habitats and to protect endangered species. Trade-off occurs in the extent to which these high-level objectives can be met simultaneously when conventions and regulations are implemented by member states. When it comes to turn the objectives into more specific goals and management actions, time and spatial scales have to be defined, and inventories of resources or components requiring protection need to be listed. During those steps the potential trade-off might appear. Also, a different perception of the trade-off may arise if objectives are framed in a socio-economic perspective rather than a merely ecological perspective. Below we illustrate potential trade-off as quantified by model results; we also provide concrete examples with contrasted levels of trade-off.

Model outputs can be used to examine the trade-off between biodiversity and fishery yields. A size-based model (LeMans) was used to calculate the relation between several community indicators and exploitation rate. Among these indicators, mean L_{max} is often used as a proxy for size composition in the community; the number of collapsed species is a proxy for loss of species richness from the community. This example shows that any level of exploitation reduced the equilibrium biomass (Figure 7.1.1). The maximum catch of all species combined would be obtained at an exploitation rate at which the mean L_{max} is reduced to 71% and total biomass to 36% of its maximum value, and 8 of 19 target species are collapsed. However, with catch at 90% of its maximum, total biomass would decline to only 58%, mean L_{max} to 85%, and only one species would be collapsed. Trade-off frontiers are commonly non-linear, such that a small reduction in one indicator (e.g. catch) corresponds to a large change in another indicator (e.g. number of collapsed species).

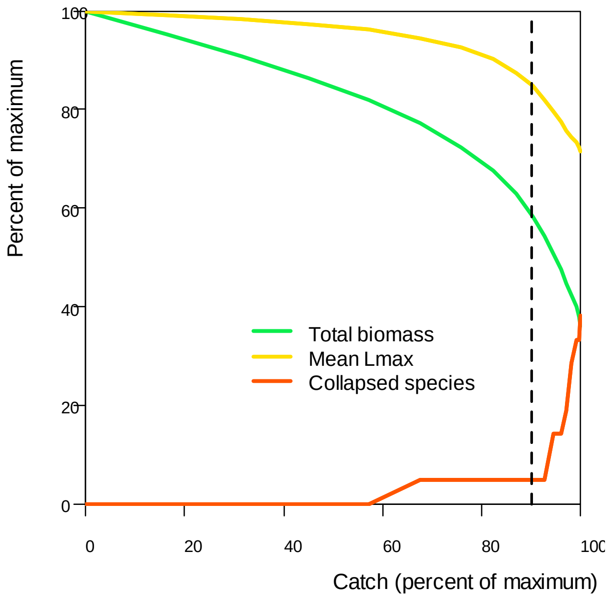

Figure 7.1.1. Trade-off between catch and community metrics in a model fish community. Catch in mass units is summed over all targeted species. Mean L_{max} refers to the average maximum length that species in the community can attain. Collapsed species are those for which stock biomass has declined to less than 10% of their unfished biomass. This size-structured model was parameterized for 19 target and two non-target species in the Georges Bank fish community. It includes size-dependent growth, maturation, predation, and fishing. All metrics were calculated over a range of exploitation rates from 0 to the value resulting in maximum catch. Adapted from Worm *et al.* (2009).

The trade-off might be less stringent when economic considerations are included. Modelling work on fisheries in Thailand by Christensen and Walters (2004) indicated that optimizing for economic profit was consistent with including ecosystem considerations, whereas optimizing landed value was in conflict with profit as well as ecosystem optimization. In the same vein, by simulating different fishery scenarios, Pitcher (2008) demonstrated that the choice of management measures is often a balancing exercise between economics and biodiversity. The results of this ecosystem-based, fisheries-policy analysis demonstrate a clear trade-off: policies that generate large revenues tend to sacrifice biodiversity, whereas policies that maintain biodiversity provide less revenue (Figure 7.1.2). Some intermediate combinations though preserve most diversity with a reasonable net present value.

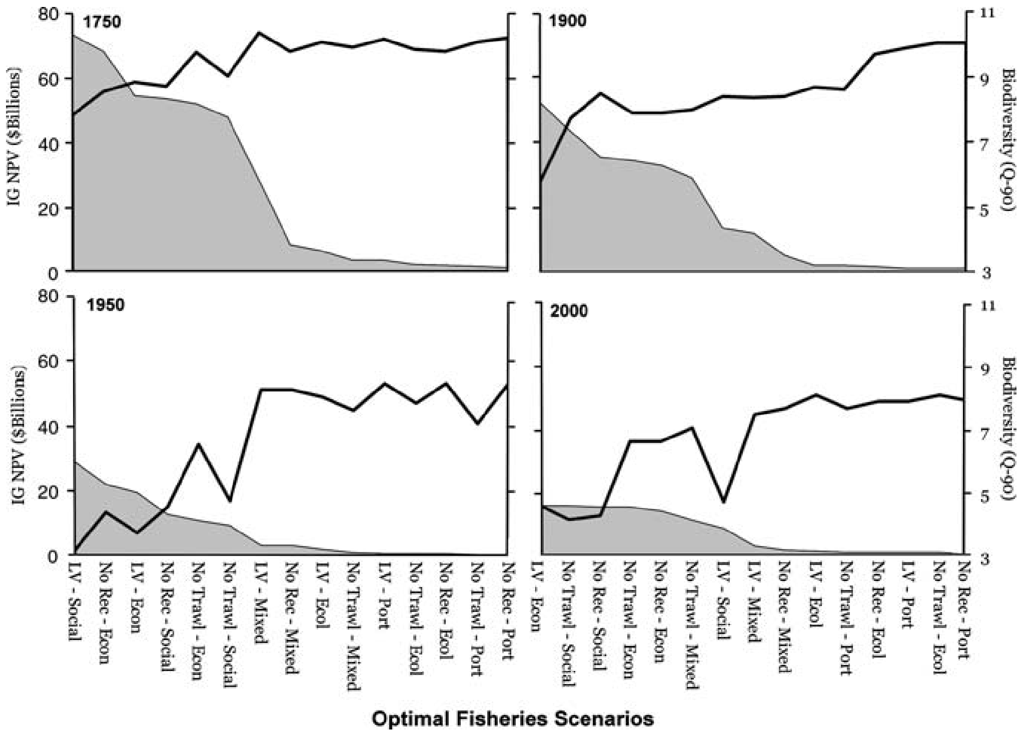

Figure. 7.1.2. Multicriteria fishery optimization using fitted ecosystem simulation models of northern British Columbia for four historic time periods fitted to all available time-series data. Left vertical axes and shaded areas show intergenerational net present value (IG NPV); right vertical axes and solid line show equilibrium biodiversity established after dynamic harvest simulations. Horizontal axes show 15 scenarios that vary harvest objectives; these are social (employment), economic (profit, NPV), ecological (longevity, life-history parameters), mixed (social, economic and ecological objectives equally weighted), and 'portfolio log-utility' (a risk averse policy, Goll and Kallsen, 2003 in Pitcher, 2008). Several fleet compositions were also included (LV = idealized 'Lost Valley' fleet; NoRec = recreational fishing gear removed; NoTrawl = bottom trawlers removed). (Modified from Pitcher, 2008).

As an example of a weak trade-off in the real world, Hutchings *et al*. (2010) examined trends in the abundance of marine fish stocks in the context of biodiversity targets. A biodiversity target adopted by the Convention on Biological Diversity (CBD) in 2002 was to reduce the rate of biodiversity loss by 2010. The authors' measure of biodiversity loss was the rate of decline of 207 fish stocks before and after 1992, the year the CBD was open for signature. The rate of biomass decline eased for 59% of populations that were declining before 1992. However, the percentage of populations below B_{MSY} remained unchanged and the rate of biomass decline increased for several top predators, many of which are below ½ B_{MSY}, which is a threshold for overfished status in the US and Australia. In this example the biodiversity and fishery targets are aligned, but the fishery target appears to be more stringent. The 2010 CBD target was a first step for recovering marine fish diversity starting from a very impacted state. The 2020 target is now completely aligned with fisheries management standards.

In deep-sea fisheries, the trade-off between sustainable use and biodiversity protection might be more explicit. Deep-sea cartilaginous fishes such as rays and chimaeras have such slow rates of population increase that recovery after depletion might require centuries; thereby sustainable exploitation of target species such as black scabbardfish, blue ling and greater forkbeard (*Adhanopus carbo*, *Molva dypterygia* and *Physis blennoides*) might not be compatible with the protection of these sensitive species (Simpfendorfer and Kyne, 2009). Deep-sea trawling was found to affect benthic community structure and biodiversity over broad spatial scales – although the soft-sediment systems of the deep ocean cover a huge area, lower levels of disturbance are required to allow recovery in the deep sea compared to those deemed sustainable in more dynamic coastal systems (Cryer *et al.*, 2002). Deep-sea trawling also threatens cold-water corals that have become an important flagship species for conservation in the deep sea (Davies *et al.*, 2007). On these grounds, conservation-oriented NGOs have made a strong case for designating deep-sea habitats as marine protected areas, or even for completely prohibiting deep-sea trawling.

7.1.4 To what extent can this trade-off be mitigated by management measures?

To meet sustainable use and conservation objectives, a combination of measures are typically required, which include catch and effort restrictions, gear modifications, and closed areas (Worm *et al.*, 2009). One measure might be more effective than others, or objectives might be attained through a combination of measures (Gavaris, 2009). Seal bycatch mitigation for instance in the Australian lobster fishery might be achieved through gear modifications, whereas spatial management of fishing effort is proposed as a suitable mitigation measure for the demersal gillnet fishery (Goldsworthy and Page, 2007). Technical gear modifications (Jennings and Revill, 2007; Donaldson *et al.*, 2010) can reduce the trade-off as selective gears or gear modifications might allow exploitation of the fish resource while protecting sensitive or endangered species and habitats. By contrast, spatial management acknowledges the trade-off and quantifies this by the relative surface of protected to unprotected areas and hence the amount of fishery yield that would be given up to conserve biodiversity.

Mitigation methods based on gear technology are mainly designed to improve size and species selectivity, and to reduce the impact on non-target species and habitats. Technically it is often possible to design fishing gears that exclude endangered, threatened or low productivity species, such as reduction of sea turtle bycatch in pelagic longline fisheries (Read, 2007) and trawl fisheries (Brewer *et al.*, 2006) or, reduction of spiny dogfish discards in hake trawls (Chosid *et al.*, 2011). Technical measures are also used more broadly to reduce fish discards (Catchpole and Gray, 2010). Several recent studies however pointed out that highly selective fishing might not be the best strategy to protect biodiversity; the removal of narrow species- or size-ranges alters the demographic composition of a population, and the species and size composition of a community, and consequently the ecosystem structure and its biodiversity (Zhou *et al.*, 2010; Garcia *et al.*, 2011; Rochet *et al.*, 2011). For example, size-based models predict that targeting a narrow size-range of larger fish would cause destabilizing trophic cascades (Andersen and Pedersen, 2010).

Fishing impact on habitats can also be mitigated through gear modifications (Valdemarsen and Suuronen, 2002; Polet *et al.*, 2010). A specific example to mitigate the physical impact is the use of modified trawl doors (e.g. He, 2007; van Marlen *et al.*, 2010). Blyth *et al.* (2004) studied the effect of restricted access by a voluntary agreement of fishermen. The effect of four different fishing regimes on benthic communities was examined, ranging from towed gears only to static gears only with intermediate combinations. The total species richness was significantly lower in sites under the two regimes with high fishing pressure by towed gears.

Spatial management makes the trade-off between biodiversity and fishery objectives explicit by specifying what fraction of harvest opportunities (e.g. percent of area closed) is given up to protect components of biodiversity (e.g. emergent benthic fauna). An effective management needs to operate on a scale consistent with the spatial scale and structure of the biodiversity component. For example, a large-scale closure will not necessarily be needed to protect smaller-scale population structure.

Several examples of spatial measures have proven to be adequate in the protection of biodiversity, in combination with its sustainable use (e.g. Blyth *et al.*, 2002; 2004). While such systems work for certain habitat-specific and non-mobile species, their utility for highly mobile stocks is questionable (Kaiser, 2005). Pichegru *et al.* (2010) however illustrated that MPAs can play an important role to protect mobile pelagic fish and top predators that rely on them, such as the African penguins (*Spheniscus demersus*).

The trade-off between fisheries and the protection of marine biodiversity might be weak if the reduction in fisheries objectives is minimal compared with the gain for conservation objectives. This is still a conjecture as methods are currently being developed to evaluate this kind of trade-off. For example, Sumaila *et al.* (2007) illustrate that a closure of 20% of the high seas may lead to the loss of only 1.8% of the current global reported marine fisheries catch (although they do not report which fraction of high seas catch would be lost). At more regional scales, it might be possible to find a balance between a minimal impact on fisheries and a maximum achievement of biodiversity objectives. Levin *et al*. (2009) examined this trade-off for a US West Coast fish assemblage and concluded that there are non-linear trade-offs between diversity (measured by species richness) and yield. The slope of the trade-off is steeper in areas of low habitat quality (Figure 7.1.3).

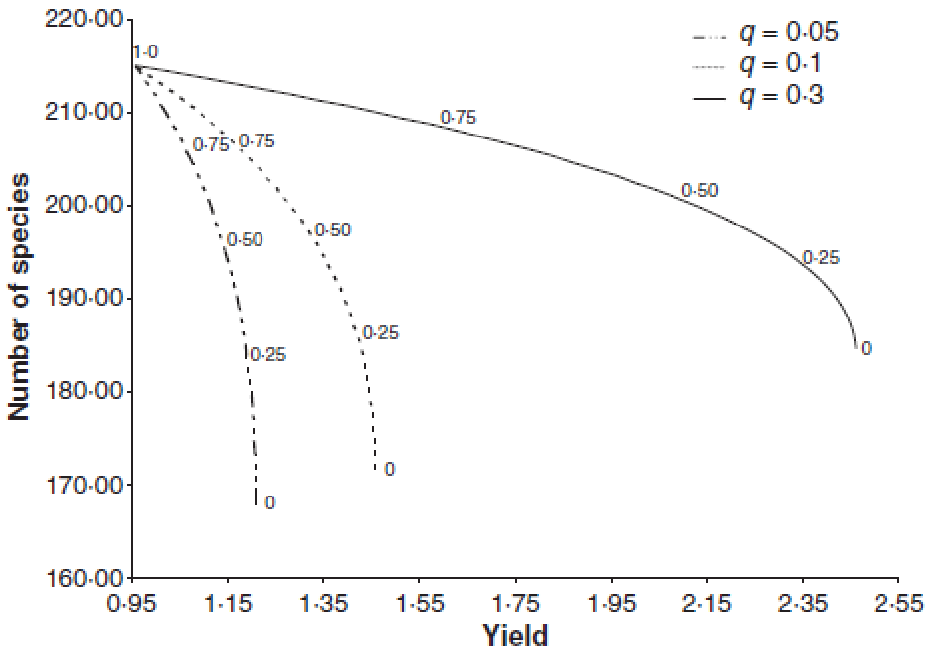

Figure 7.1.3. Biodiversity attribute (species richness) *vs*. fisheries yield for the US West Coast fish assemblage. The proportion of the coast protected by MPAs (numbers above points) vary from 0 to 1 for three ranges of habitat quality (q) in the trawled area relative to the untrawled area. Changes in species richness follow from the species-area relationship (S=16.18A$^{0.226}$) acknowledging an increase in species richness with increasing area. The proportion of fish harvested is set at the maximum sustainable yield, when there are no MPAs imposed (Modified from Levin *et al*., 2009).

7.2 Guidance to promote consistency and soundness of practices when evaluating environmental status

Biodiversity can be broadly defined as the variety, quantity and distribution of life. The breadth of this concept makes the assessment and status evaluation of biodiversity a formidable challenge. A biodiversity assessment deals not only with the most familiar resources targeted by extracting activities, or emblematic species such as marine mammals or top predators. Rare, cryptic, inconspicuous species also deserve attention in a biodiversity assessment, as do the effects of non-extractive activities such as pollution or noise that affect only sensitive species. Moreover, the complexity of a biodiversity assessment pertains not only to the diversity and amount of data to be collected to base such an assessment on the relevant evidence. The need for analysing and understanding the consequences of human activities and perturbations on biodiversity is a second daunting task. Providing a synthesis of information and knowledge pertaining to a wide diversity of biota and activities is another complication. WGECO 2010 provided general guidance on ecosystem assessment and to determine "Good Environmental Status" in the context of the Marine Strategy Framework Directive. Here we apply this framework to biodiversity and examine the issues and difficulties that are likely to arise in this specific perspective. Defining reference levels that reflect sustainable use might imply an approach different from the one associated with the objective of biodiversity conservation; this possibility is addressed in a specific subsection. An important preoccupation stems from the fact that dynamic ecosystems and changing climates will lead to continuous changes in species presence and their relative abundance within communities and ecosystems in any area. Whether reference levels should be adjusted to take these changes into account is also addressed. Finally, although much effort is directed towards providing a databased assessment of biodiversity and ecosystem status, the complexity of issues might require expert input to complement and interpret indicator-based evidence. We also examine when and how expert judgement should be used.

7.2.1 A framework for assessing biodiversity in marine ecosystems

WGECO (2010, Section 3.3; ICES 2010b) recommends a five-step process for developing an assessment of ecosystem status in the context of MSFD. Here we cite the description of each step, and discuss how it can be applied or modified to address issues specific to biodiversity.

1) An evaluation of the components of each regional ecosystem with regard to its "structure, function and processes", taking account of "natural physiographic, geographic, biological, geological and climatic factors" which identifies the parts of that particular ecosystem that are most crucial to its ecological integrity, structure, and function. In selecting these, indicators that relate to integrated aspects of the ecosystem (e.g. those that represent foodweb structure) should also be considered in order to capture the interactions of components within the regional ecosystem being assessed.

Part of this analysis can be carried out regardless of considerations specific to biodiversity, because biodiversity (e.g. at the levels of genes, species, or habitats) and its linkage to ecosystem function and resilience are characteristics of any ecosystem component so identified. This does not mean that biodiversity in each component is necessarily high. Some function or process might be represented by a single species only, highlighting the ecological importance of this species.

However, one of the functions of any natural ecosystem is simply to serve as reservoir for biodiversity (at the levels of genes and species). This function is relevant to society not only for cultural reasons, but also because it greatly facilitates adaptation of ecosystems to environmental change on time-scales ranging from the generation times of micro-organisms to macro-evolutionary times. Thus, biodiversity has societal value regardless of its current contribution to ecosystem functioning.

An extensive review of biodiversity in the ICES area, broken down by biota, has been provided by WGBIODIV (2010, Section 2; ICES 2010a).

2) An evaluation of the major human activities that are likely to result in pressures in each regional ecosystem (including physical, acoustic, chemical and biological pressures), which identifies the pressures likely to be causing the greatest perturbations within that ecosystem, and the scales on which those pressures are operating. Here we include the pressures associated with climate change since there is unequivocal evidence that humans are contributing to climate change.

Perturbation of ecosystems by anthropogenic pressures implies in most cases pressures on biodiversity. However, there are pressures that can specifically affect biodiversity, with potentially little immediate effect on other aspects of ecosystem structure and function (e.g. introduction of alien species, changes in community composition resulting from climate change). In order to effectively manage biodiversity, such pressures need to be taken into consideration.

3) Use of a scientifically peer reviewed framework (see ICES, 2006) that consists of a cross-tabulation of pressure – ecosystem component interactions that reflects which types of ecosystem components are likely to be most impacted, or otherwise be most sensitive to the pressures identified in 2, and the pressures most likely to impact detrimentally the ecosystem components identified in 1. This cross-tabulation must also link back to the potential sources of pressures (e.g. the activity-pressure relationships identified in 2).

In order to capture pressures that specifically affect biodiversity (see above), the range of "components" of such a table should include one or several categories of biodiversity, resolved by ecosystem components as necessary.

A complication arises from the fact that biodiversity can respond to the aggregated effect of multiple cumulative pressures (e.g. climate change, nutrient load, hazardous substances). In order to manage biodiversity effectively, it may be necessary to include these interactions explicitly in the matrix. In situations where deviations of biodiversity from the reference level (see below) are observed, a detailed analysis of the observed pressures and their actual impacts will narrow down the set of management measures to be taken.

4) For the components and pressures that are evaluated to be most important, ensure that one or more robust and sensitive indicators are selected. Give particular attention to the interactions between the more important components from 1 and the more severe pressures from 2, which come out of the consideration in 3.

Since biodiversity is a characteristic of any ecosystem component, but an indicator selected to reflect the overall state of a component will not necessarily reflect its inherent biodiversity, care must be taken to include dedicated biodiversity indicators for important ecosystem components. Biodiversity indicators ought to be calculated for a comprehensive range of species or taxonomic groups, including those at risk of extinction or extirpation.

The fifth and last step of the guideline and other parts of the report by WGECO (2010) address the choice of reference levels. Particulars of this step in view of biodiversity are discussed in the next section.

7.2.2 Best practices for setting reference levels that reflect sustainable use

Definitions of sustainability point to present uses that are consistent with future uses (see Section 7.1.2), which implies that present uses should maintain the ability of ecosystems to maintain and repair themselves under various levels of perturbation. However, there may be a range of conditions under which this ability is preserved. Sustainable use aims not to cross the boundaries beyond which the self-regeneration ability is impaired. Therefore the concept of sustainability is likely to be best served by limit reference points – those that define these boundaries. WGECO 2010 already pointed out that, for setting reference levels that reflect sustainable use, analyses of the relationships between perturbation and recovery would help evaluate the points at which the capacity to recover from perturbation is no longer likely to be rapid or secure – that is, limit reference points. The next section summarizes this framework and applies it to biodiversity indicators. Obviously this does not prevent to set target reference points within the sustainable area, that is, within the boundaries defined by the limit reference points. For example, those targets would achieve some optimal trade-off between various constraints. The point here is that targets are secondary to limits when the objective is sustainable use.

By contrast, conservation of biological diversity implies avoiding losses of species or components. The degree of achievement can be assessed by comparing ecosystem state to some baseline condition. OSPAR's MSFD draft advice manual on biodiversity defines a baseline as "*a specific value of state, against which subsequent values of state are compared: essentially a standard (articulated in terms of both quality and/or quantity) against which various ecological parameters can be measured*." This manual further specifies that the baseline guides the setting target reference points, although the targets generally differ from the baseline, especially in a context that allows for some degree of exploitation. Specifically, MSFD Descriptor 1 reads "*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*", which points to targets. Obviously reference limits can be added to specify how far the system is allowed to be perturbed away from the target. Reaching these limits would trigger management actions to move the system back in the target direction. In this case the limit reference levels are just additional tools to help reach the targets.

WGECO (2010, Section 6.3.2) details the complications involved in choosing reference levels that correspond to sustainable use. WGECO (2010, Section 3.3) recommends relying on, in this order of preference, (i) historical records from periods when pressures affecting the indicator were deemed sustainable, (ii) in the absence of such historical records, predicted levels of the indicator under sustainable pressure, or (iii) in the absence of reasonably reliable predictions, results of databased or modelling analyses of the relationship between the indicator and the system recovery capacity (see next section). Since measurements of biodiversity are sensitive to the methods used and theoretical or model predictions for absolute values of biodiversity indices are rare, one should also consider evidence from comparable systems and areas at other locations where pressures are considered sustainable (e.g. fully or partially protected areas, see WGECO 2010, Section 6.3.2).

As for reference levels associated with "unimpacted state", WGECO 2010 provided guidance which is supplemented by OSPAR's recent MSFD advice manual on biodiversity; we refer to those documents for a full description. Basically three approaches to setting baselines have been used.

a) Unimpacted state/negligible impacts - Baselines can be set as a state at which the anthropogenic influences on species and habitats are considered to be negligible. This state is also known as 'reference conditions'.
b) Past state - Baselines can be set as a state in the past, usually the point at which data collection on a specific species or habitat began.
c) Current state - The date of introduction of an environmental directive or policy can be used as the baseline state, typically expressed as no deterioration from this state.

Data-driven approaches to define unimpacted state have been developed for indicators of biodiversity of freshwater fish (Hermoso *et al*., 2009), mammals (Nielsen *et al*., 2007), or landscapes (Gibbons *et al*., 2008). These methods rely on large sets of samples across ranges of environmental conditions and human perturbations. Statistical models are then constructed to predict biodiversity metrics with measures of environmental conditions and human perturbations used as predictors. Reference conditions are then defined as those corresponding to low levels of human perturbation; the statistical approach accounts for variability and uncertainty by defining a reference range rather than a single reference point for a given indicator. In the marine environment, this approach might not apply to pressures such as fishing for which no area could be deemed to describe a low level of human perturbation. However, it may be applicable to habitats (descriptor 1) or benthic communities (descriptor 6) for which spatial contrast can be obtained across appropriate gradients of perturbations and environmental conditions.

Based on a defined reference condition or other baseline, OSPAR's MSFD advice manual on biodiversity identifies several target-setting options:

- Directional or trend-based targets, which might be based on direction and rate of change, or direction of change only. This will be relevant when the initial state is obviously far from the reference conditions.
- Targets set as an absolute value, either as baseline, or relative to the baseline, depending on how the baseline is defined and what deviation from it is allowed.

A typical biodiversity example of trend-based targets is the Convention on Biological Diversity (CBD) objective to reduce the rate of biodiversity loss by 2010. Another example of trend-based thresholds is provided by the International Union for Conservation of Nature categorization of threat on species, which is based on absolute thresholds for population size and geographic range, and a decline threshold on trends in population abundance.

The inconsistency between IUCN or CBD categorizations and fisheries management reference points identified in several studies (Rice and Legacé, 2007; Hutchings *et al*., 2010) might arise more from the distinction between trends and abundance levels than between conservation vs. sustainability objectives or risk-tolerance. Setting trend-based targets generally require less knowledge and quantitative bases than setting absolute values, such as the fisheries biological reference points which can be calculated after extensive data-collection and stock assessment. Proximate biodiversity targets have been based on indicator trends under the assumption that, while the indicators are well below their targets, the required direction of change is clear, even while the targets themselves are difficult to quantify. As biodiversity recovers, inconsistencies may become more apparent, for example when trends are positive but populations remain below their target levels. It is unsure however, that in future sufficient scientific knowledge will become available to set absolute reference points for biodiversity metrics especially at the community or ecosystem levels, given complexity, variability, and the lack of data on unimpacted conditions at these levels.

7.2.3 How to assess when components of biodiversity are subject to serious or irreversible harm in order to guide the setting of limits for biodiversity indicators

The overarching policy objective of the MSFD is sustainable use. WGECO 2010 detailed the conceptual link between the notion of sustainable use with a system's ability to recover rapidly and securely from the pressures applied as it is used (WGECO 2010, Section 4). WGECO therefore performed preliminary analyses of the dependence of the rate of recovery on the strength of perturbations or pressures for a selected set of empirical and model case studies, with the purpose of identifying thresholds below which recovery rates decline faster than linearly with declining system state. The high degree of uncertainty in these analyses is likely to be enhanced for the case of biodiversity. However, the body of evidence is likely to expand over the next years given the research effort devoted to biodiversity issues.

Cases in which this rationale applies have been reported by Danovaro *et al*., (2008). Investigating biodiversity–ecosystem functioning relationships for deep-sea habitats, they found that these relationships are consistently best described by exponential curves, indicating a high degree of facilitating interactions among organisms (Figure 7.2.3). Production decreases non-linearly with declining biodiversity, as required by the method proposed by WGECO (2010). When such curves would saturate at even higher biodiversity (which must be expected), the resulting inflection point would define the reference level for the respective biodiversity indicator.

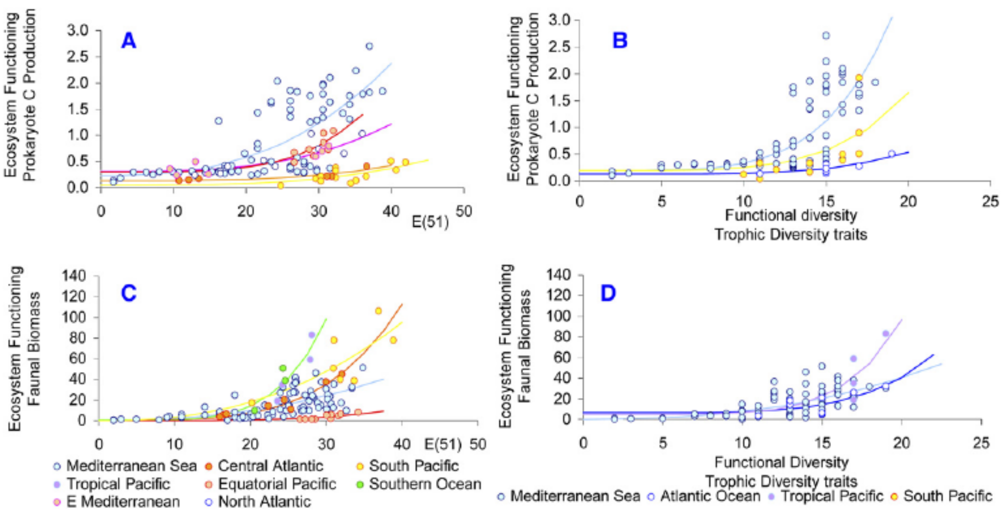

Figure 7.2.3. Relationship between biodiversity and ecosystem functioning (Danovaro *et al*., 2008). Biodiversity is (left) number of species expected for a given number of samples, or (right) number of trophic traits; ecosystem functioning is (top) prokaryote C production in mg C g^{-1} d^{-1}, or (bottom) faunal biomass, in mg C m^{-2}.

An important point to make about biodiversity is that major contribution to recovery from biodiversity loss is expected to arise from re-colonization from neighbouring habitats. Although many species depend on the presence of other species to establish themselves in a given habitat, colonization rates will generally not decline to zero even when all biodiversity in a habitat has been extirpated. Therefore the relationships between recovery rate and disturbance level is expected to differ from other types of ecosystem attributes or components, in the sense that there is potential recovery even when all biodiversity has been extirpated (e.g. Carter *et al*., 1985).

Clearly, ecosystem use that leads to global extinction of species or increases natural extinction rates is not sustainable in the sense of rapid and secure recovery, except potentially for rapidly evolving species (micro-organisms). A conceptual framework to assess a system's likelihood, speed, and ability to recover from local biodiversity loss is provided by meta-population theory (Hanski and Gaggiotti, 2004) and meta-community theory (Holyoak *et al*., 2005). A necessary, though not sufficient, condition for recovery according to these theories is good connectivity between the habitats occupied by identical species. Thus, a reference level for acceptable local loss of biodiversity might be derived by asking if the species lost can still be found in neighbouring habitats, where the neighbourhood considered depends on a species' dispersal capability. Since even in otherwise identical habitats taxonomically very different communities can establish themselves, presence of extirpated species in neighbouring habitats, however, does not necessarily mean that these will re-colonize the disturbed habitat when pressures are released.

7.2.4 Guidance on if and how expert judgment should be combined with supporting indicators to produce the best possible information and advice on ecosystem status and management options

To complement the Marine Strategy Framework Directive, the European Commission has defined criteria and methodological standards on good environmental status of marine waters (European Union 2010). Further guidance has been provided in this report, and other sources referred to above, in order to facilitate the use of structured approaches to select indicators, and ensuring that the selection of suitable reference levels is as quantitative as possible. "Best practices" should require use of such structured and formal processes to the fullest extent that the available information allows. Although the methods for selection of individual indicators and evaluation of status on the retained ones may be highly formal and structured, expert input will be required to complement, interpret and/or integrate information provided by the indicator-based evidence.

Without expert narrative to complement the quantitative information, the evidence on its own will be incomplete. Experts will be needed to input and add value to the evidence, making the fullest use of it possible, interpreting and filling gaps as necessary to have an analysis, interpretation or synthesis of information that holds together. We use "expert input" rather than "expert opinion" or "judgment" on purpose here. Expert "opinion" would suggest that the experts are not explicitly expected to take account of the indicators, rather to state what their opinions are, possibly backing up those opinions with such indicators as happen to support their opinions. Expert "judgment" may constrain the experts to have an evidence basis for their judgements, but it is still placing the experts "above" the evidence, passing "judgment" on each piece.

Instances related to biodiversity where expert input might be required include:

- Instances where the necessary data or knowledge required to set reference levels may be insufficient; in such cases expert input would be required to set reference levels based on knowledge of the particular system in question and on indicator values and/or reference points potentially existing in other ecosystems or situations.
- Alternatively, when there is no quantitative basis for choosing a reference point, expert input might be needed to qualitatively appraise the status *e.g.* relative to Good Environmental Status, for a given indicator in a particular context. This differs from the first case in being more qualitative.
- Expert input is required to combine and integrate information from several indicators. There are several levels of integration required to move from evaluation of the individual indicators to an assessment of overall status, *e.g.* Good Environmental Status: indicators within individual Attributes of a Descriptor; status across all the Attributes within a Descriptor; and status across all Descriptors. To combine results over even a moderate number of indicators, rigorously and prescriptively "algorithmic" approaches to produce "a number" for overall status from scores on all the individual indicators is not feasible nor even desirable. This was highlighted by WGECO 2010. At the integration stage, status on different indicators must be weighted, uncertainties across indicators combined, and interactions taken into account. Various multivariate analytical tools exist to aid in pattern extraction from multiple indicators. These only address issues of correlations (redundancies) among indicators. They do not address questions about the inherent information that the individual indicators may contain about actual ecosystem status, nor the importance that should be given to indicators that may be outside (or far on the positive side) of their respective reference levels. Although it is theoretically possible to calibrate patterns and trends across multiple indicators, doing so requires adequate replication and controlled conditions for calibration. What is needed is a synthesis, not a reduction, of the information in the indicators, and this can be achieved by expert input.
- When there is no quantitative model that formally links management options to expected impacts on biodiversity, expert input will be required to perform a qualitative risk assessment and provide advice on management options based on the state and pressure indicator values.

Even in formalized frameworks, expert input is known to depend on the background, level of information and level of detail required in the assessment, and is prone to subjectivity and value judgement (Rochet and Rice, 2005; Piet *et al.*, 2008). The following practices might limit these drawbacks to the extent possible.

1) The group of experts conducting the synthesis has to include experts with a variety of perspectives, not just disciplines. For example when it comes to marine biodiversity, experts may come from backgrounds along a continuum from strict conservation to sustainable use perspectives. A suitable group should include experts from a fisheries science background, and from a conservation biology background.
2) The expert group should be encouraged to lay out the framework for its analysis, detailing the different steps and elements to be considered in making their evaluations and such criteria that it will be using in assembling its conclusions. This inventory step is required before the evaluation

itself uses whatever information is available for analysis, to avoid the framework being determined by results the experts may prefer (or dislike).

3) The experts should seek out consensus relative to this framework, but they should not seek forced compromises of different interpretations just to have a single story. Reporting about "variance in interpretation" in the narrative will contribute to informing the reader about the uncertainty of the assessment. When several interpretations are competing and no agreement can be found about which one is the most likely, all interpretations should be reported as possible realities, and the implications of each explained.

4) In addition, the experts should provide a qualitative measure of uncertainty in their conclusions, which indicates their confidence in the conclusion and the kind of knowledge added to the available quantitative information (Halpern *et al*., 2007).

5) The expert group should annotate, comment, and provide explanation for its conclusions, including an explicit link between the background information used and the outcomes.

7.2.5 Best practices for setting reference points in changing conditions (to ensure sound science and avoid shifting baselines)

Setting reference levels of indicators relative to baseline conditions generally involves explicit or implicit assumptions about background values of demographic parameters and community composition under prevailing environmental conditions. When these background values change it may become necessary and appropriate to adjust the reference level in response. The properties of dynamic feedback control systems are well understood for single-species management (Walters, 1986). Whether or not reference levels should be adjusted in response to changing conditions depends on the nature of the change, the degree of mechanistic understanding between the changing conditions and the indicator, and the level of unexplained variability (Walters and Parma, 1996). One category of changing conditions is exogenous climate variability and monotonic climate change. In general, if the change is short-term variability around the mean, reference levels should not be changed because conditions will average out over the time-scale of management actions and there is a strong risk of chasing unexplained noise in the system. If the variability is longer term or a monotonic change, it may be advantageous to adjust the reference level in response to changing conditions provided a high level of confidence in the link between the changing conditions and the indicator and a low level of unexplained variability from other sources. In the case of monotonic changes in an external driver, it needs to be remembered that the link between the indicator and the driver may change at some threshold level of the driver.

In an example from single-species management (Brunel, 2009), knowledge of the relationship between an environmental (climate) variable(s) and an ecosystem attribute is used to modify reference levels such the ability to keep a fish stock "within Safe Biological Limits", SBL), is improved. For commercial stocks policy aims to keep the stock within SBL, which are represented by reference values for spawning–stock biomass (SSB) and the level of exploitation (i.e. fishing mortality, F). Management is based on measures of the state of the stock in relation to their associated (limit, precautionary or sustainable) reference levels. Adjusting the reference points according to the prevailing environmental conditions is one way to incorporate environmental information in stock management. Environmental variability is responsible for long-term trends in recruitment of fish (Brunel and Boucher, 2007) or of sudden switches from one regime to another (Alheit, 2005). Adaptive management through environment-driven changes in F reference points is one way to tune the level of exploitation to the current level of productivity of the stock (ICES, 2006).

This approach requires a known relationship between the environmental variable and a factor (recruitment) that determines one of the indicators (SSB) of the stock that determines whether or not it is within SBL (Figure 7.2.5). In response to the state of the environment, as reflected by the environmental variable, the reference level of another indicator of the stock (the exploitation level F) is adjusted. A comparison of the performance of management applying conventional reference levels to that based on reference levels varying according to the state of the environment showed that the latter performed slightly better overall but considerably better when a detrimental change in the environment occurred. The benefits of using the environmental reference levels were the greatest for those stocks with the strongest environment–recruitment relationship.

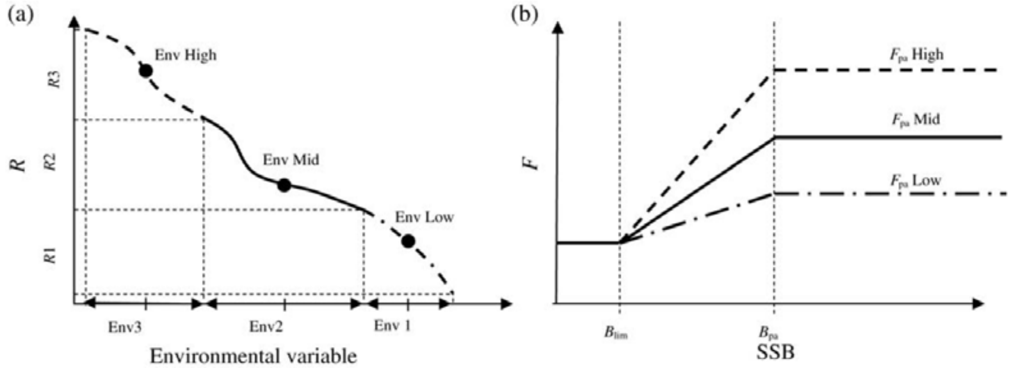

Figure 7.2.5. Management incorporating environmental change. (a) The response function of recruitment to environment (illustrative example) is used to define three possible states of the environment. (b) Three different SSB–F relationships can be used to set the F reference point according to SSB, depending on the state of the environment. Adapted from Brunel *et al.*, 2009.

This single-species example shows that adjusting reference levels in response to environmental variability can help to achieve policy objectives. In principle a similar approach could be used to modify the reference level of a biodiversity attribute/indicator in response to changing environmental conditions, should the necessary information become available.

Another category of change occurs when the reference level of one indicator is conditional on the value of another indicator, for example when the reference level of a given species depends on the abundance of an interacting species. In this case, there may be a formula for adjusting the reference level, but the same conditions apply—that the strength of the interaction is well understood relative to unexplained sources of variability. A third case involves proximate human-induced changes, for example a reduction in the habitat of a species of concern. In such a case, it may be pragmatic to change the reference level if it is no longer attainable. On the other hand, shifting the baseline for setting reference levels may weaken conservation standards. In any of these cases, the performance of the adaptive feedback response should be tested with intensive simulation that includes all the main sources of uncertainty.

In a decision-making context, establishing an adaptive feedback mechanism for adjusting reference levels provides more flexibility compared with setting reference levels that are fixed in time. Such an adaptive process may reduce initial resistance to setting reference levels and increase stakeholder buy-in. On the other hand, there is a risk of reference levels being changed inappropriately (e.g. chasing noise) or of particular user groups subverting the process to benefit their interests.

To date there is very little experience about how to adapt the reference levels of biodiversity indicators in response to changing conditions. A hypothetical example might involve a reference level of species diversity for a benthic community subject to human disturbance (e.g. pollution, bottom fishing). If species diversity in the benthic community is also changing due to colonization by non-endemic species, should the reference level for disturbance be adjusted in response? There is no general answer to this question. In the first place, the choice of appropriate metrics has not been completely resolved. Second, there are few apparent threshold levels of biodiversity indicators that can be used as a basis for reference levels (Section 6.2.3). Given the challenge in identifying metrics and reference levels for biodiversity, it seems unlikely that the criteria (derived from single-species experience) for adjusting them in response to changing conditions would be met.

7.3 Summary and recommendations

WGECO examined the components of the Strategic Initiative on Biodiversity that relate to the ecosystem effects of fishing activities. The high-level objectives of international and regional seas conventions on biodiversity are consistent with the corresponding fishery conventions. For example, the Convention on Biological Diversity recognizes the sustainable use of biodiversity components. In turn, the Code of Conduct for Responsible Fisheries advises measures to conserve biodiversity of aquatic habitats and to protect endangered species. Inconsistencies become more apparent when the terms of the high-level objectives are defined and the objectives are implemented as guidelines and descriptors. Any level of harvest will affect the size structure, species composition, and biomass of the community, impacts that will be reflected in biodiversity indicators. Thus, any management strategy will involve some level of trade-off between biodiversity and sustainable use.

WGECO considered several examples of this trade-off for model and empirical results. The trade-off frontier is typically curvilinear, such that a small reduction in sustainable yield may translate into a larger increase in biodiversity indicators. Simulation studies have been used to identify harvest policies that preserve most of the biodiversity while maintaining most of the value of the fishery. In the case of depleted communities, the short–term conservation and sustainable-use objectives are strongly aligned when both prescribe rebuilding; it is only after rebuilding occurs that incompatibilities may become more explicit.

The trade-off between biodiversity conservation and sustainable use can be mitigated, not eliminated, with the choice of management measures. Certain gear types are known to have greater ecosystem impacts than others. Gear technology is used to improve size and species selectivity, and to reduce the impact on non-target species and habitats. However, selective fishing may not be the best strategy to protect biodiversity if it alters size composition and community structure. Spatial management involves an explicit trade-off between fishing opportunities and the protection of habitats and other components of biodiversity. This trade-off is particularly strong for deep-sea fisheries and other habitats with slow growing and/or fragile fauna.

Quantifying this trade-off and the benefits of spatial management requires knowledge of the degree of overlap between fisheries and vulnerable habitats and species, and valuation of the costs and benefits.

WGECO 2010 articulated a process for assessing ecosystem status in the context of the Marine Strategy Framework Directive. WGECO concludes that this five-step process can be applied, in particular, to assess biodiversity indicators in the context of the Strategic Biodiversity Initiative. Regarding best practices for setting reference levels that reflect sustainable use, WGECO noted that reference levels for sustainable use are generally expressed as limits, whereas biodiversity goals are generally expressed as target levels or the desired direction of trends. It is unsure however, that sufficient scientific knowledge will become available to set absolute reference points for biodiversity metrics, given complexity, variability, and the lack of data on unimpacted conditions. Another basis may be needed to set biodiversity targets. It should be possible to set these biodiversity targets within the limits of sustainable use. There might be circumstances requiring the setting of reference points that are consistent across indicators associated with different objectives. The MSFD requires a consistent definition of "Good Environmental Status" and the MSFD descriptors are oriented towards both biodiversity conservation and sustainable use. In this case the aim is to identify the overlap between the tolerated loss from the biodiversity targets and the boundaries associated with sustainable use. Reference points should be identified within this overlap area. Inconsistencies can occur also between trends in biodiversity indicators and limit reference points for sustainable use.

WGECO 2010 detailed the conceptual link between the notion of sustainable use with a system's ability to recover rapidly and securely from pressures that are applied. This procedure relates a measure of recovery (e.g. productivity, recruitment) to the state variable or indicator and looks for non-linearities in this relationship. In principle, this procedure can be applied to assess when components of biodiversity are subject to irreversible harm, but there are few examples in which recovery capacity has been measured in relation to biodiversity. Moreover, the framework has been developed for single pressures and should be further developed as knowledge of cumulative impacts of multiple pressures increases. Research to be done in anticipation of the 2018 MSFD review could be to collect new data of this type and to synthesize existing data, possibly in a meta-analytic framework.

WGECO identified several situations in which expert input is needed to identify reference levels and to determine the status of biodiversity indicators or ecosystems. WGECO recommend the use a structured procedure for obtaining and documenting expert input and outline such a procedure relevant in the context of the MSFD.

WGECO recognizes that it may be seen as necessary to adjust the reference levels of biodiversity indicators in response to changing conditions, especially climate variation. Procedures for adaptively changing reference levels are well understood for single-species management, but the criteria for adaptive management are unlikely to be met in a biodiversity context. Given the difficulty of identifying reference levels for biodiversity, WGECO does not recommend a procedure for adaptively changing reference points at this time.

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8 ToR F: Marine spatial planning, human pressures and biodiversity

This ToR is in three parts;

- WGECO takes note of the report of WKCMSP and takes some of the recommendations and thoughts forward.
- WGECO provides a view on the link between changes in both human pressures and biodiversity in relation to marine spatial planning. The focus lies on the large development plans for offshore renewables and on relevant information for the development of pressure indicators in relation to biodiversity indicators such as habitat biodiversity.
- WGECO identifies some of the general gaps in spatial data and analysis to support area based management regimes such as marine spatial planning.

In the first part the conclusions and recommendations of the report of WKCMSP are assessed and scientifically relevant points are elaborated further.

The second part discusses the development of marine integrated management using marine spatial planning from a scientific perspective highlighting some challenges. Some of which are the implementation of risk-based decision-making and the quantification of uncertainty in a planning processes. Risk-based decision-making in spatial planning is related to the importance of providing the science base for activity-pressure-state relationships. Previous work of WGECO focused on the use of the concept of the activity-pressure-state relationship integrated assessments including the provision of guidelines to develop indicators or assess pressures. Here its use is discussed in relation to the development process of marine spatial plans together with its relevance in the definition of operational objectives for planning. Based on previous work of WGECO (ICES 2010) some generic pressures related to offshore renewable energy are listed together with a mini review on development plans for offshore renewable in Denmark, Germany and UK. Changes in human pressures at different scales and related changes in habitat biodiversity are described with a hypothetical example to provide information on how to derive related pressure indicators.

The third part identifies some of the gaps in the availability of spatially resolved data and its application to MSP. Recommendations are made how ICES can improve the data provision to support MSP and build on existing data infrastructure. WGECO recommends that these data are made more widely available and promoted to relevant stakeholders.

In summary, WGECO identifies a number of topics and subjects as being of specific relevance to ICES. A common theme of these topics is that they deal with the bigger picture and focus on methods, transboundary/regional questions and the need for frameworks to deal with logistics and on MSP on a regional/transnational scale, i.e. a theme that reflects the nature and role of ICES as a whole. However, the role of ICES to support area based management such as MSP is limited to the provision of specific scientific advice and the provision of spatially resolved data. As already outlined in the WKCMSP report the planning process involved in MSP is based on interaction between policy, managers and stakeholders, with the scientific community having a data provision function. It is the view of WGECO that ICES could support MSP processes through a number of useful activities. In addition to the suggestions identified in the WKCMSP report, WGECO recommends:

- using experience and networks to facilitate regional assessments, focusing on dealing with MSP in transboundary/regional seas contexts;
- providing a science base for activity-pressure-state relationships to support risk based decision-making in planning processes;
- evaluation of ecosystem services, which can then be assessed in relation to trade-offs within MSP processes;
- evaluating concepts such as carrying capacity in relation to the acceptable degree of change in the state of habitat biodiversity due to pressures from large renewable energy developments. This includes the assessment of local impacts and the extrapolation to larger scales in the absence of empirical data;
- WGECO recognizes that a number of groups and initiatives have struggled to develop methods for assessing cumulative or combined impacts that take account of both additive, synergistic and antagonistic effects. However, this remains an important knowledge gap for MSP;
- WCECO recognizes a potential for ICES to provide a service of the provision of spatially resolved information relevant to area based management.

8.1 Moving beyond the WKCMSP report

The Workshop on the Science for area-based management: Coastal and Marine Spatial Planning (WKCMSP) in practice aimed to establish the current scientific state of knowledge within ICES on CMSP and to identify gaps and scientific limitations to support CMSP. WGECO takes note of this report and takes some of the recommendations and thoughts forward.

The WKCMSP report (ICES 2011) captures the majority of the main issues and data/research gaps surrounding the practical implementation of MSP. Many of the points described in the report represent some of the main challenges facing scientists and managers today and WGECO sees a number of areas where ICES, with the regional and ecological focus and the networks and logistical capacity of the organization, is in a unique position to make a substantial contribution to MSP science, development and evaluation. However, ICES should also acknowledge that WKCMSP is not a starting point for MSP science in general and should ensure that any initiatives are of added value in relation to already mature research and policy processes, and that planned activities are aligned with the expected needs of society and the research communities that are already developing the science related to MSP and related fields. In the following some issues are outlined in more detail.

The assessment of cumulative impacts is a general demand in the MSP process and it was identified within the report as one of the key scientific gaps. WGECO recognizes that the concept of cumulative impacts is not well defined in the growing literature of practical assessments (see e.g. Halpern *et al*., 2008; Ban *et al*., 2010; Stelzenmüller *et al*., 2010). Cumulative impacts are sometimes described as the sum of the number of impacts observed, while the term is also used to describe the combined impact of multiple pressures over space and time (McDonald *et al*., 2007), the latter interpretation taking account of interactions between impacts that may have synergistic, antagonistic or additive effects. Antagonism is a cumulative impact value lower than the sum of individual impacts, and synergy is a value greater than the sum of individual impacts (Folt *et al*., 1999). In this report, WGECO has adopted the latter interpretation of the term “cumulative”.

A recent report by HELCOM (2010) presented a well documented approach that, based on the Halpern *et al*. (2008) method, assesses human pressures and impacts on the Baltic Sea marine environment. However, the work describes additive pressures, i.e. it does not take account of any interactions between these pressures. In practice this may result in the development of management measures in the Baltic Sea that do not address the true nature of pressures at sea. There is a clear gap in the scientific knowledge base on how to detect and evaluate such interactions between pressures, despite of the fact that Darling and Côté (2008) conclude that synergistic effects generally are more common than additive ones. The lack of clear separation and definitions of what is meant by cumulative impacts calls for the development of practical guidance for their assessment within MSP.

Often the availability of necessary spatial data is not aligned with ongoing spatial planning initiatives. This is partly due to the planning process, where often scientists, stakeholders and planners are not involved at the same time, but rather consulted when demand arises. WGECO recognizes the potential role of ICES to improve the coordination of provision of spatially resolved data (see the third part of this ToR).

8.2 Marine spatial planning and the link between changes of human pressures and biodiversity

8.2.1 A scientific perspective on the development of integrated marine management using marine spatial planning

Place-based or spatial management such as marine spatial planning (MSP) is seen to facilitate an ecosystem based management (Lackey, 1998). MSP is an integrated planning framework that informs the spatial distribution of activities in the ocean in order to support current and future uses of ocean ecosystems and maintain the delivery of valuable ecosystem services for future generations in a way that meets ecological, economic and social objectives (Foley *et al*., 2010). Thus one of the strengths of MSP is the ability to integrate the management of a diverse range of human activities and hence their associated pressures to achieve the higher-level objectives of healthy ecosystems, sustainable use, and the delivery of ecosystem goods and services.

Single sectors and their spatial use conflict with other sectors or marine conservation measures have driven most marine spatial planning initiatives around the world (Dover, 2008). In practice, MSP initiatives that are based on strategic environmental assessments generally result in specialized technical management approaches that lack the environmental context in terms of its contribution to cumulative effects (ICES 2010). Thus there are only a few examples such as the Large Ocean Management Plans in Canada (www.dfo-mpo.gc.ca) where ecologically and biologically significant areas have been defined together with social, cultural, and economic overviews for several oceans and coastal management areas. As a means to validate risk-based decision-making, a compendium of ecosystem vulnerabilities, geospatial analysis tools, a definition of ecosystem zone of influence and regional vulnerabilities profiles are being piloted in relevant coastal zones.

In general, risk based decision-making is a process that organizes information about the possibility for one or more unwanted outcomes to occur into a broad, orderly structure that helps decision-makers make more informed management choices. WGECO recognizes the need to provide the science base such as activity-pressure-state relationships into risk-based decision-making in spatial planning processes.

Often one of the main obstacles with regard to integrated marine management is the lack of relevant knowledge, information, and data. However, data and knowledge are never complete at the beginning of a MSP process (Douvere and Ehler, 2010). Thanks to recent advances in spatially explicit tools for mapping and visualization of the distribution of pressures those outputs can be provided to planners (e.g. Eastwood *et al*., 2007, Ban *et al*., 2010). As a marine spatial plan describes the spatial and temporal allocation of resource use, it is crucial to assess the uncertainty associated with the data used. Beyond the issue of incorporating uncertainty in a decision-making process and accounting for the accumulation of the latter it is also critical to visualize the uncertainty associated with the outcomes of possible spatial management scenarios (Stelzenmüller *et al*., 2010). Thus the development of spatial management scenarios to support marine planning requires a spatially explicit framework that incorporates various sources of uncertainty. For instance, Walker *et al*. (2003) described three sources of uncertainty in any model-based decision support tool. Uncertainty can be related to location (where the uncertainty occurs in the model complex), level (where the uncertainty occurs on the gradient between knowledge and ignorance), and nature (whether uncertainty is due to knowledge gaps or to the variability inherent in the system). Thus the quantification of uncertainty is an important element in a risk-based decision framework.

8.2.2 The use of activity-pressure-state relationships in marine spatial planning

The development of marine spatial planning (MSP) has been identified as an important tool for the delivery of ecosystem based management (EBM; Douvere, 2008). Inherent within the context of MSP is the need for a clear understanding of the relationship between activities, pressures and environmental state (condition), and the spatial distribution of environmental components. Pressures arising from human activities in the marine environment have been identified (e.g. offshore renewables; WGECO 2010) but the relationship between pressures and state is less clear. Changes in state depend on the type of habitat, nature, extent and frequency of the pressure, and the resilience (recovery time) of its characteristic species (Robinson *et al*., 2008).

High level objectives focus on the state of the system however management is targeted at the activities and hence there is a critical need to understand how changes in activity levels brought about by management will result in changes in the state. The total level of pressure is a combination of the pressure arising from each of the sectors. One of the key strengths of MSP is that it provides a framework to combine these effects. Human activities are the only aspect of these ecosystems that can be managed and traditionally management has been applied on a sector by sector basis. Therefore the emphasis of MSP should be to integrate spatial management of sectors to take account of cumulative impacts of their activities.

Current EU projects are examining cumulative impacts of human activities in the marine environment and the relationship between activity-pressure-state including: MESMA (Monitoring and Evaluation of Spatially Managed Areas, www.mesma.org) in the context of spatially managed areas; and ODEMM (Operational Development of Ecosystem Marine Management, www.liv.ac.uk/odemm) in the context of the Marine Strategy Framework Directive (MSFD). Drawing upon these projects, Figure 8.2.2.1 presents an adapted framework that may be suitable for adoption for MSP within the context of EBM. This framework would allow integration of activity-pressure-state relationships from the different sectors to inform area based management through MSP to achieve high level objectives.

The framework acknowledges that linkages between components (human and environmental) are interrelated and multi-directional. State of ecosystem components and the ecosystem goods and services (EGSs) they provide is affected both directly and indirectly by environmental, socio-cultural and economic components. The framework is dynamic and MSP should take account of changes in the strength of contribution to pressures over time. Guidance on the MSP process (e.g. selection, mapping and assessment of ecosystem components and indicators related to operational objectives, evaluation of management effectiveness, adaptative management; MESMA project) could be used to compliment this framework. Emphasis through MSP should be on appropriate spatial management of sectors to take account of cumulative activities and activities. We have developed an example case study of large renewable energy developments to consider potential pressures and associated impacts of these pressures on habitat biodiversity.

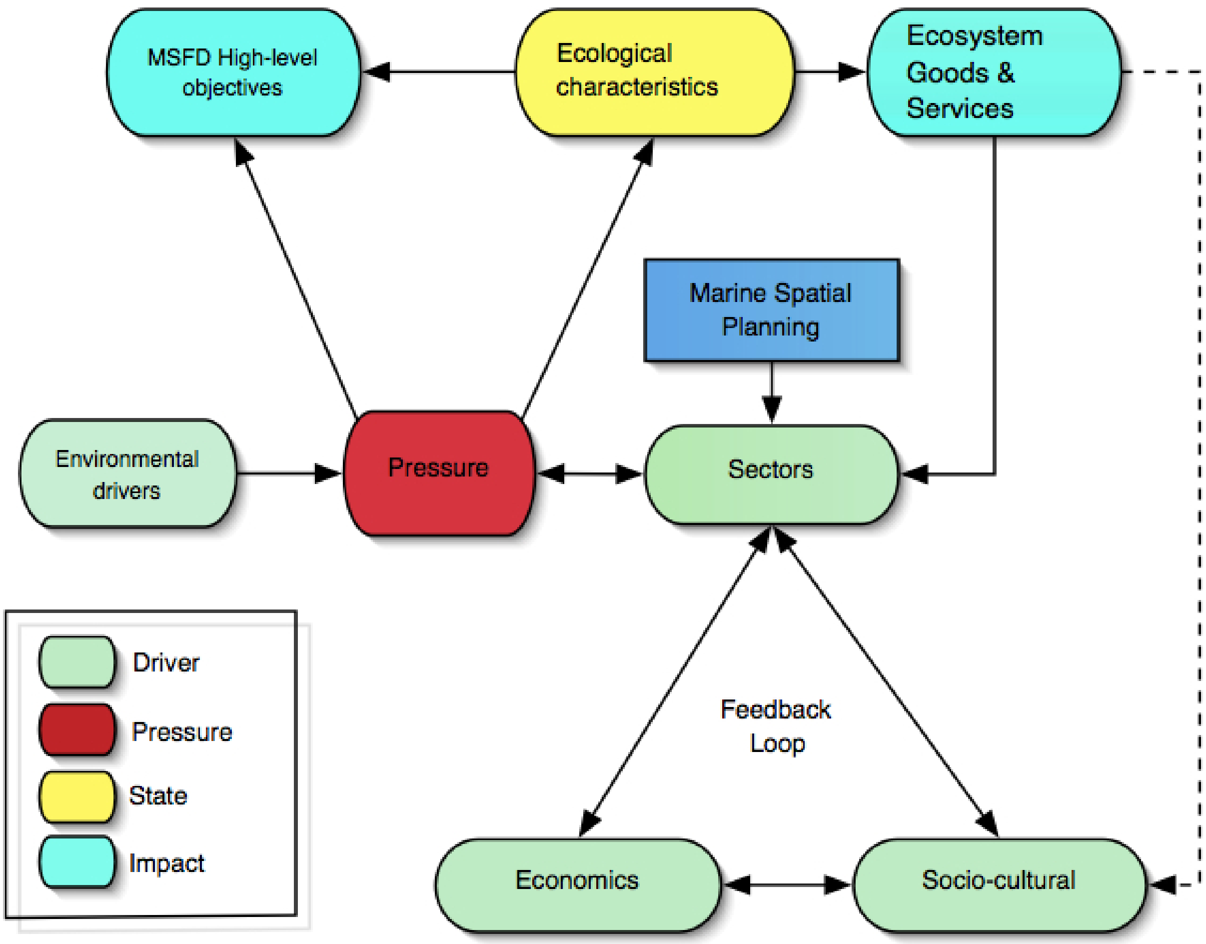


Figure 8.2.2.1. Potential framework for marine spatial planning in the context of ecosystem based management drawing upon the ODEMM Linkage Framework (EU FP7 Project, Grant number 244273). High level objectives (e.g. healthy ecosystems and sustainable use) focus on the state of the system and feed in through MSP (blue box); management measures focus on the sectors (human activities).

8.2.3 The future challenge of large renewable development plans

In recent years the rate of development of renewable energy projects has increased. The major policy driver for development of the renewable energy sector is the EU Renewable Energy Directive (2009) which sets targets for all Member States to reduce their greenhouse gas emissions to meet the Community's greenhouse gas emission reduction commitments up to 2020. Across the EU, targets are to meet 20% of energy needs from renewable sources by 2020.

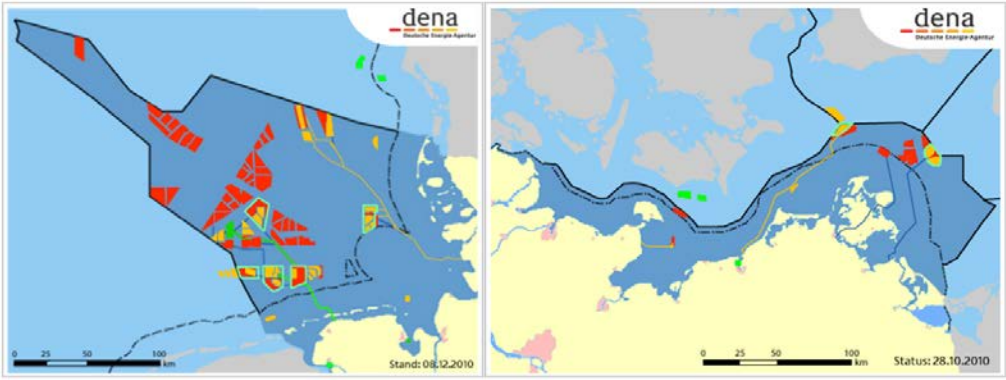
Renewable energy developments will continue to increase in number as Member States work to meet individual targets. The scale of proposals, particularly in the offshore wind sector, in terms of size (number of turbines, total area) and energy outputs as a result of technological developments is also increasing. In the following, we provide examples of existing and proposed large renewable energy developments in Denmark, Germany and the UK.

8.2.3.1 Denmark

Denmark is currently developing its offshore renewable energy capacity. A 400 MW windfarm (111 turbines) will be built in 2012–2013 to the west of the island of Anholt in the Skagerrak. As a part of the Danish government's long-term plan to meet EU obligations by 2020 and a CO2 neutral energy supply by 2050 (Danish Government 2011), a 600 MW windfarm on Kriegers Flak in the Baltic Sea has been put to tender as well as 400 MW of smaller windfarms closer to the coast. Denmark currently has a number of wind farms installed including Horns Rev I and II.

8.2.3.2 Germany

Under the overall management of the Federal Ministry for the Environment, Nature Conservation and Reactor Safety (BMU), the German government has developed a strategy on the use of offshore wind energy that takes nature conservation and other interests into account. According to this, the installation of 20 to 25 GW offshore capacity is seen as possible by 2030. The maritime spatial plan for the German EEZ North and Baltic Sea is legally binding and contains designated sectorial preference areas (BMVBS, 2009). The plan identifies low-conflict areas which could be considered suitable for offshore wind energy installations. These areas were classified as special areas suitable for wind energy after an appraisal of the various interests on the basis of the Marine Facilities Ordinance. However, most of the application areas for wind energy development are outside those preference areas. Further, applications for wind farms within the 12 nm zone have been submitted to the states of Lower Saxony, Schleswig-Holstein and Mecklenburg-Vorpommern. Currently, the Alpha Ventus wind farm is the world's most distant (45 km off the coast) and deepest located (30 m water depth) wind farm (Figure 8.2.3.2.1, indicated as online).



Online, Approved, Under consideration

Figure 8.2.3.2.1. German development plans of offshore wind energy in the North Sea (left) and Baltic Sea (right) as of December 2010. Taken from www.offshore-wind.de.

8.2.3.3 UK

In December 2008, the UK agreed to a legally binding UK target through the EU Renewable Energy Directive to provide 15% of its energy from renewable energy by 2020 (DECC 2009). There is commitment to expand the long-term incentives for large renewable electricity developments and recent analysis indicates that up to 30% of the UK's electricity could be provided by renewables (up from 5.5% in 2009); it is envisaged that on and offshore wind would account for more than two-thirds of this production (HM Government 2009). This target is even higher for some of the Devolved Administrations, for example the Scottish Government recently increased the target level for renewable electricity generation from 50% to 80% of gross electricity consumption in Scotland by 2020 (Scottish Government 2010). A recently completed Strategic Environmental Assessment concluded that 25 GW of offshore wind development would be permissible in UK waters, in addition to existing plans for 8 GW of offshore wind (HM Government 2009).

The most significant individual offshore renewable project in UK waters is the proposed Dogger Bank wind farm development in the North Sea, located between 125 and 190 km off the Yorkshire coast (Figure 8.2.3.3.1). This zone is the largest within the Crown Estates third licensing round for UK offshore wind and covers an area of approximately 8660 km² with its outer limit aligned to UK continental shelf limit (Crown Estate). The consortium (Forewind, www.forewind.co.uk) developing the zone has agreed a target installed capacity of 9 GW, although the site has the capacity for up to 13 GW of electricity generation. The consents process (surveys, assessments and planning) is currently underway, and is expected to be completed in 2014 (www.thecrownestate.co.uk/offshore_wind_energy).

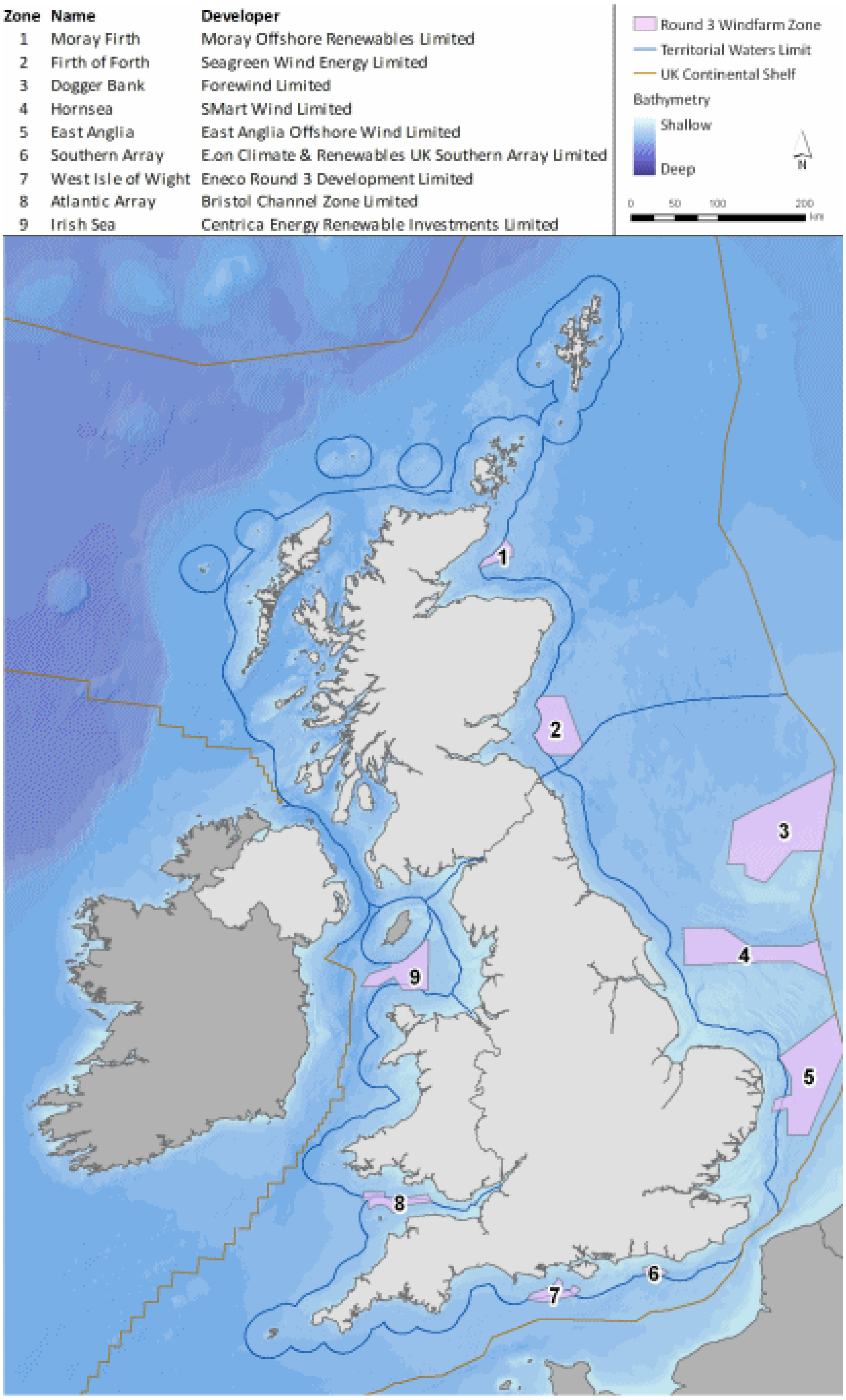


Figure 8.2.3.3.1 Map of Round 3 Offshore wind farms in UK waters (Source: Crown Estate, http://www.thecrownestate.co.uk/70-interactive-maps-r3.htm?txtName=Map-R3-Zone-3).

8.2.3.4 Summary of key issues identified for large renewable developments

The country examples above illustrate the number and scale of some of the proposed large renewable energy developments in the marine environment. All have the potential to apply pressures on marine components and associated changes in state. Key issues identified (e.g. ICES 2010; IPC 2010) for consideration for large-scale proposals include:

- Scale of the proposal (unprecedented size);

- Transboundary impacts (consultation required with other Member States);
- Cumulative impacts with other developments in the area;
- Ecological impacts (construction and operation);
- Construction noise impacts;
- Socio-economic impacts (fishing and other uses/users); and
- Archaeology (disturbance to known/unknown archaeological sites).

8.2.3.5 Generic pressures of large renewable energy development and changes in biodiversity: a hypothetical case study

Large renewable developments may exert a number of pressures on the marine environment during construction and operational phases. The main ecosystem components likely to be affected are birds, marine mammals, marine habitats and their characteristic species, functions and processes. Whilst some pressures are common to all offshore renewable activities, e.g. the potential for smothering, substratum loss, siltation and underwater noise during construction, other pressures are activity and phase (construction or operational) specific (see ICES 2010; Table 8.2.3.5.1). The intention here is not to provide a comprehensive assessment of the pressures from larger renewable energy developments which could be drawn from the wider literature (e.g. Inger *et al.*, 2009; Grecian *et al.*, 2010; ICES 2010; ODEMM, www.liv.ac.uk/odemm) but to provide an indicative list of potential pressures, whose associated activities could be spatially managed through MSP.

Table 8.2.3.5.1. Indicative list of potential generic pressures associated with offshore renewable activities (wind, WF; wave, W; tidal, T); pressures are common to all activities unless indicated. This table draws upon other research (e.g. ICES 2010; ODEMM) but is not a comprehensive list of pressures. Note that the nature, extent or frequency of pressure is not inferred.

	Phase	
Pressure type	Construction	Operational
Physical	Smothering Substrata loss (or change) Siltation Abrasion Underwater noise	Siltation (W, T) Underwater noise
Hydrological		Changes in water flow rate
Biological	Death/injury through collision	Barrier to species movement Death/injury through collision
Other	Introduction of synthetic and non-synthetic compounds	Electro-magnetic changes (WF, W) Salinity changes (T)

The main ecosystem components likely to be affected by the pressures resulting from renewable energy plans are birds, marine mammals, marine habitats and the related functions and processes. Thus a degree of change in habitat biodiversity will be one of the main results of these uses. Physical habitats respond to human pressures in a fundamentally different way to the biotic components of the system. *In extremis* a biological species can be extirpated, with consequences for community composition and biodiversity measures, but in general affected biotic components retain an ability to recover (resilience).

Using a hypothetical example we will illustrate the complexity of trying to link pressures and changes in habitat biodiversity. While an area of habitat x may be described as having been damaged or removed due to pressures resulting from human activities, the same area of the marine environment exists but the area formerly occupied by habitat x will now be occupied by habitat y. Depending on the pressure the transition from habitat x to y may be a continuum, and might be seen as a decrease in quality of habitat x to some point where it transitions to habitat y. For example, if a sandflat has a set of structures built on it, they may cause silt deposition. Initially this could be referred to as a decrease in the quality of the sand habitat but this could also be considered as a new area of muddy-sand habitat. The former habitat (sandflat) might have been dominated by small worms and amphipods, whereas the muddy sand becomes colonized by lugworms (*Arenicola*) and brown shrimp (*Crangon*). From this example it is clear that the system has changed, there has been a decrease in one habitat and an increase in another. The wider impacts of the change in state are unknown.

If the overarching objective of management is to conserve a particular habitat, or a particular proportion of each habitat, then an appropriate metric is the area of habitat. However, implementation of the objectives requires identification of the habitat type and quality required. It is possible to envisage a monitoring programme measuring habitat area, type and quality on a routine basis. However, it may be more appropriate to think in terms of infrequent assessment of habitat distributions and regular monitoring of pressures. If one understands the activity-pressure-state relationship, which for physical habitat may be relatively straightforward, this provides a clear link to potential degree and location of habitat change.

National plans for the development of large renewable energy developments (Section 8.2.3) demonstrate that an increase of pressure can be expected due to an increase in activities. However, changes in the state of habitat biodiversity due to the cumulative pressures at larger spatial scales (e.g. the southern North Sea) are not clear. Thus calculations of the carrying capacity consistent with the acceptable level of change in habitats may be applied on different spatial scales. However, in the light of the above examples the question raised is can concepts be used as a basis in risk based decision-making at the scale of national plans or even regional spatial management initiatives.

Thus for any given spatial planning initiative the amount of change in pressures and related impacts on the habitat is regulated to meet the defined operational objectives. High-level management goals need to be translated into operational objectives to allow the elaboration of specific targets, limits and measures. Operational objectives are defined as those for which specific, measureable, achievable, realistic and time limited (SMART) targets can be set such that management measures can be fitted and performance can be evaluated. Operational objectives aim to implement the overall goal of the spatial plan such as e.g. the promotion of offshore renewables.

In summary, WGECO highlights the need to develop and incorporate activity-pressure-state relationships in operational objectives for MSP to support risk-based decision-making. Based on the activity-pressure-state relationship, pressure indicators in relation to renewable energy developments incorporate the magnitude, spatial extent and frequency of the underlying activities. WGECO recognizes that the acceptable level of change in such pressure indicators and the related combined impacts on biodiversity indicators such as habitat biodiversity may be able to be estimated locally based on current scientific knowledge. However, the extent to which known activity-pressure-state relationships at a local scale can be extrapolated to a regional level is not clear. Integrated marine management using MSP facilitates a holistic assessment of activity-pressure-state relationships within a planning area.

8.3 Identification of gaps in spatial data and analyses to support area based management

WGECO found it to be beyond the scope of the meeting to provide a thorough metadatabase of spatially resolved data to support MSP. Instead, WGECO has identified gaps in availability and analysis of spatially resolved data which ICES has the potential to develop to support MSP. This work is based on the other tasks within ToR f e.g. thoroughly reading the WKCMSP report, exploring links between activity, pressures and state and application of some of these thoughts in light of the case study on renewable energy development. In identifying gaps it is important for ICES to note that the marine planning arena brings together a wide spectrum of disciplines and the MSFD has made certain types of data (e.g. commercial fish and fisheries) relevant to a wider range of clients and stakeholders. As a result, outputs which might be perceived as simple or obvious to traditional ICES audiences may be considered extremely useful by these new audiences.

Gaps in availability and analyses of spatially resolved data are identified in Tables 8.3.1 and 8.3.2. The tables comment on the scope and provision of such data and suggest the degree to which ICES could contribute to their development. Most of the gaps identified have relevance for the CFP, MSFD, N2K and MSP. These policies and processes all require spatially resolved data related to ecological components and processes as well as the distribution of activities and associated pressures.

The importance of having data at the appropriate scale is recognized. The resolution of data required for MSP will depend on the scale for which spatial management is applied. For instance, fishing effort data at the scale of ICES statistical rectangles would be appropriate to analyses on a regional or subregional scale, while the resolution might be too low to be useful for spatial planning in smaller areas. Temporal scale is also of importance, as the particular natural processes and human activities may vary greatly over time and at different rates. Estimation of uncertainty in relation to spatially resolved data is a key gap in data from MSP.

Many of the gaps identified by WGECO are related to further development and new analyses of data held within ICES DataCentre. WGECO recommends that these data integrated more effectively and made available and disseminated in appropriate formats useful to data customers, and promoted to relevant stakeholders. ICES could establish the links between data needs of MSP and the data collected via the DCF with the purpose of highlighting needs for further integration.

Table 8.3.1. Gaps in availability of spatially resolved data identified by WGECO.

Category of spatially resolved DATA	Subcategory	Comments/scope for ICES
Fish habitats	Spawning & nursery areas	ICES could produce species "fact sheets" and downloadable GIS files to provide state-of-the-art spatial information on the spawning and nursery areas of fish species at the highest possible spatial resolution. Novel approaches may need to be explored to improve existing data, incl. gathering of knowledge from fishing sector, etc. Numerous ICES WGs have previously dealt with mapping of spawning and nursery areas for different species and in different marine areas, e.g. SGRESP (Petitgas, 2010). This SGRESP approach is useful to MSP and ICES could make valuable contribiutions to MSP by expanding this work to other species and making products readily available online (i.e. as files independent of reports, GIS files, etc.).
	Location of known fish habitat types incl. pelagic habitats	Biological characterization/definition and mapping of common habitat types of commercial fish species in the ICES area, combined with maps of known distribution of such habitats (e.g. from regional/national mapping projects and initiatives) would be a valuable planning tool for integrated spatial management in relation to MSFD. In particular such products would be useful in planning of MSFD protected area site selection (MSFD Art. 13, 4–6). Such an analysis could include pelagic habitats (e.g. Planque *et al*., 2006) which may be relevant in development of MSP, vertical spatial zonation, definition of biological characteristics e.g. reproductive volume (MacKenzie *et al*., 2000).
	Migratory pathways	Mapping of known migratory pathways of marine species would be useful and is often an overlooked component in marine planning on a larger scale. Such maps would require novel approaches to analysing existing data, gathering of knowledge from fisheries stakeholders, etc.
Species	Commercial fish and shellfish species	State-of-the-art species distribution maps for commercial fish species in the ICES area are currently available in the ICES FishMap website http://www.ices.dk/marineworld/ices-fishmap.asp. This online tool could be expanded to include information on different life stages of focal species, providing the planning process with spatial information on stock structure, etc. It should be the aim to provide maps at the highest possible resolution. Examples of such mapping initiatives can be seen in the following websites http://sharpfin.nmfs.noaa.gov/website/EFH_Mapper/map.aspx and http://www.nero.noaa.gov/hcd/index2a.htm. In addition, numerous ICES WGs have dealt with mapping of commercial fish species, e.g. SGRESP (Petitgas, 2010).
	Sensitive species	Maps showing the known spatial distribution of species that may be considered "sensitive" in marine ecosystems (e.g. in the context of CFP, red lists, etc). The identification of relevant species can be extracted from ICES WG reports.

Category of spatially resolved DATA	Subcategory	Comments/scope for ICES
	Key species (foodweb)	Maps showing the known spatial distribution of species that may be considered "key" in marine ecosystems (e.g. species that perform key functions as top predators, prey species, etc). The identification of relevant species can be extracted from ICES WG reports. Such spatial data would be of relevant to MSP in light of MSFD. The ICES website (www.ices.dk) is currently home to the EcoSystemData Online Warehouse, which provides spatially resolved data on fish stomach sampling, contaminants in sediments, aggregated trawl-survey data, etc. This website could be expanded to include key and sensitive species and habitats and other spatial layers. ICES should make planners more aware of the availability of these data sources. ICES currently publishes zooplankton status reports (e.g. O'Brien *et al*., 2008) which could be made available as downloadable GIS files to provide spatially resolved data for use in analyses of primary and secondary production.
Physical / Ecological processes	Spatio-temporal resolution of upwellings, fronts, etc.	Fronts, upwellings, etc. can be characterized/identified/modelled and mapped by relevant WGs based on existing data. Such spatial maps would be useful in planning for identifying ecological units, potential productivity hot spots, etc. ICES is currently involved in these areas e.g. WGOOFE oceanography group and ICES Reports on Ocean Climate.
	Productivity hot spots	Areas of high productivity can be characterized/identified/ modelled and mapped by relevant WGs. Maps could have relevance for sectoral spatial planning (e.g. identifying valuable fishing grounds, conservation planning).
	Biodiversity hot spots	Areas of high biological diversity can be characterized/identified/ modelled and mapped by relevant WGs. Maps could have relevance for MSP in general and for conservation planning in particular.
Anthropogenic activities	Fishing activity by vessels <15m (w/out VMS)	High resolution spatial data are currently available for fishing effort of vessels larger than 15 m. However, only low resolution data or very few data are available for smaller vessels. Since fishing vessels without VMS data make up the majority of the fleet in some countries, it would be of great value for MSP if ICES could develop methods/standards/services for making existing spatial data for smaller vessels as useful as possible and make those data available in highest possible resolution.
	Discard observations	Develop methods and maps to make discard observations more useable in spatial management. In order to make such data as ecologically relevant as possible, mapping of total catch (landings and discards) should distinguish between areas where species are extracted (mortality) and areas where discards are dumped (food supply). Mapping and analysis of discarding patterns in EU fisheries is one of the objectives of the EU project BADMINTON (http://83.212.243.10/badminton.html).

Table 8.3.2. Gaps in availability of spatially resolved analyses identified by WGECO.

Category of spatially resolved ANALYSES	Subcategory	Comments/scope for ICES
	Development of system of smaller statistical units within ICES	ICES should aim to present data at higher spatial resolution as existing ICES statistical units are too large for use in planning within smaller marine areas. WGECO recommends that appropriate spatial scale requires further discussion within the ICES community (see also ICES 2008).
	Spatial overlap analysis of fish habitatvs.existing MPAs, spatial plans	Maps and analyses of the spatial overlap between fish habitats and existing protected areas (e.g. Natura 2000) are relevant in MSP and MPA designation. These analyses can help to optimize spatial allocation of activities or marine conservation areas (e.g. MPAs).
	Spatial extent of fishing activities	Develop aggregated maps of fishing activities (and associated pressure footprints and intensity) using modelling approached (incorporating point data) to provide spatial coverage at appropriate resolution. These data should be aggregated at métier, species or temporal scales.

8.4 References

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Annex 2: Agenda

1000 Wednesday 13 April

Plenary

Introductions

Presentation on using ICES SharePoint/Printer and other services

Overview of meeting work plan **Dave Reid**

Presentation on WGECO approach to **ToR a**: Provide guidance on the use of the proportion of large fish indicator in areas outside the North Sea. **Sam Shepherd and Simon Greenstreet**

Presentation on WGECO approach to **ToR b**: Review the use of science in the development and implementation of "integrated ecosystem management plans" (IEMPs) including objectives setting and performance evaluation as well as other considerations. **Jake Rice, Ellen Kenchington and Mark Tasker**

Presentation on WGECO approach to **ToR c**: Review and comment on the SGMPAN report which presents general guidelines for MPA network design processes that anticipate the effects of climate change on marine ecosystems. **Chris Frid**

Presentation on WGECO approach to **ToR d**: ToR on the European Marine Strategy Directive. **Stuart Rogers and Dave Reid**

- Identify elements of the WGECO work that may help determine status for the 11 Descriptors set out in the Commission;
- Provide views on what good environmental status (GES) might be for those descriptors, including methods that could be used to determine status.

Presentation on WGECO approach to **ToR e**: ToR in relation to the Strategic Initiative on Biodiversity that is being developed by Simon Jennings and Mark Tasker. **Marie-Joelle Rochet and Jeremy Collie.**

Presentation on WGECO approach to new **ToR f**: Take note of and comment on the Report of the Workshop on the Science for area-based management: Coastal and Marine Spatial Planning in Practice (WKCMSP) - **Vanessa and Ellen Pecceu**

- provide information that could be used in setting pressure indicators that would complement biodiversity indicators currently being developed by the Strategic Initiative on Biodiversity Advice and Science (SIBAS). Particular consideration should be given to assessing the impacts of very large renewable energy plans with a view to identifying/predicting potentially catastrophic outcomes;
- identify spatially resolved data, for e.g. spawning grounds, fishery activity, habitats, etc.

Getting the show on the road

Allocation of people to ToR

Discussion groups for ToRs a–f;

Uploading material to SharePoint, etc.

0900–1000 Thursday 14 April

Meeting of ToR leaders to inform each other of direction each group is taking.

1100–1200 Plenary for any emerging issues

0900 Friday 15 April

Discussion groups for all ToRs

***** Meeting to follow a format of break-out group and plenary discussion as required with times to be posted daily based on progress*****

Weekend: WGECO works through both Saturday and Sunday with a later start on Saturday and a late day plenary on Sunday.

Tuesday 19 April

The last plenary session will be scheduled for the afternoon. Remaining time will be spent tidying up the report, finalizing references, etc. Each ToR group should identify at least one member who will be present Tuesday afternoon to do this. There will be no formal meeting on the Wednesday, as this is very close to Easter, and I anticipate a lot of early leavers!!!!!

Annex 3: WGECO terms of reference for the next meeting

The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by David Reid, Ireland, will meet in Copenhagen, Denmark, xx–xx XXX 2012 to:

b) Review the use of science in the development and implementation of "integrated ecosystem management plans" (IEMPs) including objectives setting and performance evaluation as well as other considerations.

WGECO will report by DATE to the attention of the Advisory Committee.

Supporting Information

Priority	The current activities of this Group will lead ICES into issues related to the ecosystem affects of fisheries, especially with regard to the application of the Precautionary Approach. Consequently, these activities are considered to have a very high priority.
Scientific justification	Term of Reference a) Several countries are conducting or have recently completed significant studies in this area and the subject would benefit from a review of progress and an evaluation of the results obtained. The last review of significant studies occurred in 1996 by ICES Study Group on Unaccounted Mortalities. A review of more recent work will determine the need for revision and update on planning and methodology for studying this subject. Term of Reference b) All fishing activities have influences that extend beyond removing target species. The approach recommended by FAO is that responsible fisheries technology should achieve management objectives with a minimum of side effects and that they should be subject to ongoing review. WGFTFB members and others are currently undertaking a range of research programmes to provide the means to minimize side effects.
Resource requirements	The research programmes which provide the main input to this group are already underway, and resources are already committed. The additional resource required to undertake additional activities in the framework of this group is negligible.
Participants	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities	None.
Financial	No financial implications.
Linkages to advisory committees	There are no obvious direct linkages with the advisory committees.
Linkages to other committees or groups	There is a very close working relationship with all the groups of the Fisheries Technology Committee. It is also very relevant to the Working Group on Ecosystem Effects of Fisheries.
Linkages to other organizations	The work of this group is closely aligned with similar work in FAO and in the Census of Marine Life Programme.