ANNEX 1: Biodiversity components: species & habitat lists

The following lists of species and habitats (embedded files) contain an agreed list habitat and species used by HBDSEGB in the development of targets and indicators for GES. These were developed via ICG-COBAM, and thereby provide a level of consistency in the assessment of biodiversity across the North-East Atlantic region.

The lists contain:

- Predominant habitats and functional groups of species
- Special (Listed) species and habitats from Community legislation and international agreements.
- Additional species being considered within some subregions for potential use to represent the broader functional group in which they occur. This selection is guided by the criteria below and is an ongoing process.

The lists are intended as a common starting point for identification of indicators for GES. These lists may be extended to include an agreed subset of more common species, representative for the functional groups (in liaison with ICG-COBAM, under OSPAR). However, the agreed functional species groups contained in the attached version can already be regarded as guidance for species assessments under MSFD.

The following guidance on the selection criteria for species within each functional group (from ICG-COBAM (1) 11/4/1) provides a clear view on the operability (practicability) and effectiveness of indicators based on the suggested species. The selection of species to be assessed under MSFD in the OSPAR maritime area should be representative in terms of:

i. their abundance and distribution (i.e. also naturally predominant species as well as species that are predominant as an effect of human activities should be included);
ii. their sensitivity towards specific human activities;
iii. their suitability for the respective indicators and descriptors of the EU COM decision;
iv. the practicability (incl. cost effectiveness) to monitor them;
v. their inclusion in existing monitoring programmes and time-series data;
vi. their association with specific habitats.

MSFD Habitats list:

MSFD Draft Species list (under development):
ANNEX 2 Supporting information for the Benthic Habitats Chapter 3

Annex 2A: Distribution of Benthic Habitats throughout UK waters.

Subtidal and deep-sea habitat types are derived principally from modelling; intertidal habitat types are derived from survey data but cannot be seen on this scale of map.

Map A – Potential benthic habitat distribution in the UK, based on EU SeaMap modelled data (2010). The 18 predominant habitats have been merged into 8 broad types for ease of visualisation (different reef types and sediment types are not highlighted). Littoral (intertidal) habitats are not shown.
Map B – Potential Annex I Habitats Directive Habitat distribution in the UK. The map shows the potential distribution of three Annex I habitat types that occur away from the coast. Other Annex I habitats are not shown.
Map C – Distribution of OSPAR Threatened and Declining habitats (biogenic reefs and seamounts)

Map D – Distribution of OSPAR Threatened and Declining habitats (all other [non-reef] habitats)
There are clear regional differences in the distribution of benthic habitats within UK waters, although information on the distribution of offshore habitats (especially subtidal rock) is still incomplete. Intertidal rocky habitats (including rocky and boulder shores and sea cliffs) are widespread, occurring in all Regional Seas. Notable exceptions are the south-eastern and north-western coasts of England, as well as parts of Wales, where intertidal sediments form extensive beaches, sandbanks, saltmarshes and muddy shorelines. In other areas (e.g. Scotland and Northern Ireland) such stretches of intertidal sediments are often interrupted by rocky promontories and headlands.

The largest known areas of subtidal rock (including biogenic reefs) occur in Scottish waters, particularly to the west of the Hebrides and around Shetland, though some extensive areas also occur off Devon and Cornwall. Elsewhere this habitat occurs mainly as a narrow band adjacent to rocky shores. Biogenic reefs are built by marine species such as horse mussels (*Modiolus modiolus*) found mainly to the north, and ross worms (*Sabellaria spinulosa*), which are more common in the south and east.

Subtidal sediments cover the vast majority of the continental shelf around the UK. Most of the shelf is covered by sands, gravels or mixed sediments, with muds mainly accumulating in deep basins in the Northern North Sea and Irish Sea, as well as in sheltered sealochs in Scotland and Northern Ireland; each of these sediment types supports distinctive communities. For MSFD purposes, they have been divided into shallow and shelf subtidal sediment types, the distinction being that shallow sediments may be regularly disturbed by surface waves and therefore support quite different communities to shelf sediments. Large expanses of shallow subtidal sediments are particularly widespread in the Irish Sea, the Eastern Channel and the Southern North Sea and occur out to considerable distances offshore. Sediments within coastal lagoons are largely confined to southern England and western Scotland. Conversely, shelf sediments occur much closer to coasts where the water deepens rapidly, e.g. around most of Scotland, Northern Ireland, Cornwall and on Rockall Bank, west of Scotland.

Deep-sea habitats occur below 200m, and are found beyond the continental shelf edge. Within UK waters they mainly occur to the north and west of Scotland and west of Rockall, although there are also small areas in the extreme southwest off Cornwall. Most of these are sediment habitats, with rocky habitats and reefs largely confined to seamounts and similar structures.
Annex 2B Relationship between predominant habitats, and Special (listed) habitats and EUNIS habitat classes.

The table shows:

a. The predominant habitats (based on the TG1 types; Cochrane et al. 2010) and the Special (listed) habitats and benthic species (from the Habitats Directive and OSPAR Convention) that are associated with each predominant habitat;
b. The relationships between the predominant types and the listed types.
c. The relationships between the MSFD habitats and the EUNIS habitat classes;

The regions (~MSFD subregions) indicate where each habitat/species occurs (green = pretty likely/certain, ? = possible).

Note that these regional sea boundaries are slightly different the ones used to undertake the habitat assessment for Charting Progress 2 due to improvements in the resolution of the biogeographic data used to draw the boundaries between regions.
Annex 2D Summary of the possible baseline-setting and target-setting approaches

Further information on baseline-setting under the Habitats Directive (HD)

Range and area under the Habitats Directive require the setting of ‘favourable reference values’ (FRVs) which effectively act as baselines against which the FCS targets are set. These values are identified on the basis of habitat ‘viability’, which is a difficult concept to apply to marine habitats. The favourable reference value for range and area must be at least that when the Directive came into force. Information on historic distribution may be used when defining the favourable reference range and area, and ‘best expert judgement’ may be used to define it in the absence of other data. For many Member States, including the UK, FCS is largely determined by the status of habitats at the time the Habitats Directive came into force nationally (1994), and the use of historical data is minimal. This means the baseline used under the Habitats Directive is essentially ‘current state’ (see figure above), and the opportunity for recovery of habitats that were extinct or extirpated (in a region) or significantly modified before 1994 is limited (for example, European oyster beds disappeared in the North Sea before 1994 and have not been considered in the FCS assessments for Annex I Reef under the Directive).
Annex 2E Sensitivity matrix and pressure thresholds

The matrix of vulnerability (below) shows the likely impact of a pressure on a habitat. Impact (vulnerability) can be determined by combining information on sensitivity and exposure. The scores for ‘sensitivity’ and ‘exposure to pressures’ are multiplied to derive a coarse grading for feature vulnerability. This grading is set out in the Table entitled ‘Categories of vulnerability’.

The matrix of vulnerability. The figures presented are for illustrative purposes only.

<table>
<thead>
<tr>
<th>Relative exposure of the habitat to a specific pressure</th>
<th>Relative sensitivity of the habitat to a specific pressure</th>
</tr>
</thead>
<tbody>
<tr>
<td>High (3)</td>
<td>Moderate (2)</td>
</tr>
<tr>
<td>High (3)</td>
<td>9</td>
</tr>
<tr>
<td>Medium (2)</td>
<td>6</td>
</tr>
<tr>
<td>Low (1)</td>
<td>3</td>
</tr>
<tr>
<td>None (0)</td>
<td>0</td>
</tr>
</tbody>
</table>

Note the level of likely impact (vulnerability) will always be categorised ‘insufficient information to make any assessment’ in cases where there is inadequate information to assess either the exposure OR sensitivity of a given feature.

Categories of vulnerability. The figures presented are for illustrative purposes only.

- High vulnerability: 6 to 9
- Moderate vulnerability: 3 to 5
- Low vulnerability: 1 to 2
- Vulnerability identified, but not quantified as level of exposure unknown.
- No known vulnerability: 0
- Insufficient information to make any assessment

The pressure thresholds (targets) set out in spreadsheets below have been developed by:

- Determining the sensitivity of the MSFD habitat using information from the MCZ project work
- Using the ‘Matrix of vulnerability’ to determine what level of pressure exposure the MSFD habitat could tolerate in line with ‘moderate vulnerability’, given its specific sensitivity to that pressure (i.e. up to a score of 5 or a light blue colour).
- Setting a ‘qualitative pressure exposure threshold’ in line with a maximum vulnerability of ‘moderate’ for each habitat. The threshold categories are:
  - No or low exposure to pressure;
  - Up to moderate exposure to pressure;
- Up to high exposure to pressure;
- Not Exposed;
- Unknown

- Setting a confidence score (in brackets) in line with the confidence score assigned to the sensitivity assessment (i.e. if the confidence in the sensitivity assessment is low, the confidence in the qualitative pressure exposure threshold will also be low).

The ‘qualitative pressure thresholds/targets’ in spreadsheet ‘Sensitivities matrix and pressure targets’ can be articulated in terms of both the temporal frequency and spatial distribution of a given pressure benchmark (see below for more information about pressure benchmarks). This is because in some cases it may be appropriate to manage the temporal frequency of a pressure to achieve GES, in some cases its spatial distribution and in some cases both. For example, many biogenic reefs are significantly impacted by the first occurrence of physical abrasion, and therefore managing its temporal frequency may not be as effective as managing its spatial distribution.

The matrix below presents:

i) an assessment of the sensitivity of 108 MCZ features (which have been grouped into Broadscale Habitats (based on EUNIS Classification Level 3), Habitats of Conservation Interest and Species of Conservation Interest) to 5 physical and biological pressures that can be linked to human activities in the marine environment

ii) an assessment of the sensitivity of the MSFD habitats to these 5 pressures, established though a ‘translation’ of MCZ broadscale and listed habitat sensitivities into MSFD Predominant and listed habitat sensitivities. Note that sensitivity scores have not been generated for Habitats Directive Annex I habitats, as these were not covered by the MCZ project.

iii) a set of pressure thresholds based on the sensitivity scores of all the MSFD habitats (except Habitats Directive Annex I habitats) for these 5 pressures

The final recommended pressure indicators and targets (thresholds) are set out in Annex 7 of this report (as well as in Chapter 3).

Annex 2F  Background to sensitivity matrix information

Below is an extra from the Natural England & JNCC guidance for use of sensitivity matrix information published in January 2011 for the Marine Conservation Zone Project.

Sensitivity of marine species and habitats: Guidance on using the Sensitivity Matrix, Pressures and Activities Matrix, and combination tables to predict potential implications of MCZ designation.

This guidance is to help in the use and interpretation of the sensitivity matrices and tables that have been developed by Defra, Natural England and JNCC, and supplied to the regional MCZ projects. The information for estimating general sensitivity of marine features includes the feature sensitivity to pressures matrix, developed by ABPmer, and the activities/features tables subsequently put together by Natural England. The feature sensitivity to pressures matrix shows the relative sensitivities of marine features to environmental pressures, at specified benchmarks. The activities/features tables build on this information, and allow looking up of the sensitivity of each marine species and habitat feature to a given pressure, whilst simultaneously being able to see which activities are associated with that pressure. This was achieved by combining the data of the feature sensitivity to pressures matrix and an activities and pressures association matrix, produced by JNCC.

Specific issues to note when interpreting the tables and matrices

1. Interpretation of sensitivity assessments and associated human activities

Due to the way that the activities/features tables are put together there is a chance that the results could be misinterpreted. For example, if the table shows that a feature is highly sensitive to a particular pressure and that that pressure is associated with a particular activity; it does not automatically follow that the activity would have to be prohibited in order to protect the feature at a given location. A discussion and/or evaluation will need to be made, taking into account the way activities operate, to ensure the pressure or pressures are actually occurring on the specific sites. Conversely, if a feature was deemed to have a medium or low sensitivity to a pressure, it would not necessarily mean that activities associated with the pressure could be maintained at current levels, particularly if an associated activity was causing the pressure at or above the pressure benchmark (see below). Again, this will depend on how activities are operating within the site, for example frequency, gear type etc.

2. Pressure benchmarks

The sensitivity assessments were made using pressure benchmarks, which defined a particular level of pressure. They considered what the effect on the feature would be if the pressure occurred at that pressure benchmark, or level. For example, the sensitivity of horse mussel beds to ‘shallow abrasion’ was determined to be high, where the Natural England & JNCC guidance for use of sensitivity matrix information – January 2011 pressure benchmark for shallow abrasion is ‘damage to seabed surface and penetration ≤25mm’.

However, while there are activities associated with causing shallow abrasion, it may not be the case that those activities necessarily cause the pressure at that benchmark, within a specific marine site. Therefore, the benchmark describes the level of pressure to which the feature is sensitive, and not the level, or intensity of the associated activity(s).

Intensity and type of activity, and therefore site-level sensitivity considerations, will be location and activity-specific so definitive general assessments are difficult to make accurately. Decisions on
management will ultimately require expert judgement on a case-by-case basis, and the evidence on which any decision is based should be provided for transparency.

3. Sensitivity ‘ranges’

In some cases the sensitivity assessment of certain features to a particular pressure is necessarily presented as a range (e.g. ‘medium to high sensitivity’). This might be because the feature in question is broad-scale, and comprised of component sub-habits that are not all equally sensitive to the same pressure. An example might be subtidal sand, which includes both high and low mobility habitats. It’s sensitivity to ‘physical removal and extraction of the substratum’ is considered to range from ‘Low’ to ‘High’ sensitivity on the basis that stable diverse communities will exist in some areas (higher sensitivity) whilst mobile and less diverse areas will exist in others (lower sensitivity).

In these cases, for the purposes of clarity, the precautionary approach has been taken and the higher sensitivity result has been listed in the matrix and the collation table. However, where the sensitivity assessment is a range, this has been indicated by an asterisk and it will be possible to refer to the original information to work out the detail of the assessment if required. Further information, if available, such as the presence of sub-habits or species, could be used to refine the sensitivity assessment for a particular broad-scale habitat site.

4. Features

The feature sensitivity to pressures matrix will be used by a number of different marine projects across the UK. As such the features listed are not specific to the MCZ Project or the Ecological Network Guidance. However, the broad-scale habitats and features of conservation importance listed in the Ecological Network Guidance are all represented in the features sensitivity to pressures matrix.

5. Coincident features and activities

In the over-arching sensitivity table there will be instances where it appears that a feature is sensitive to an activity that does not overlap with the feature. This is a factor of combining the sensitivity matrix and pressure/activity matrix, linked through the pressures; and common sense will need to be applied in these cases to disregard those activities that, although they may be associated with the pressure, do not interact with the feature. For example, coastal saltmarsh is highly sensitive to ‘physical change (to another seabed type)’, which is in turn associated with fishing through hydraulic dredging. However, fishing through hydraulic dredge would obviously not occur on saltmarsh. Later in the process, the Natural England & JNCC guidance for use of sensitivity matrix information – January 2011 analyses of ‘exposure of features to pressures’ should resolve this issue. Natural England and JNCC will also look into ways to screen out these anomalous results in the meantime.

6. ‘Compatibility’ of activities

The matrices and activities/features tables do not show ‘compatibility’ of activities with features. The compatibility or incompatibility of features with activities will depend on a wide range of site-specific variables, such as location, intensity (frequency and duration), and current management of activity. Using a matrix approach for predicting ‘compatibility’ in this simplified way would give spurious and in many cases misleading answers. The activities/features tables provide an initial indication of which activities are associated with pressures that can impact certain features.

This is the first step in the process of assessing exposure, and although the regional MCZ projects will undertake a vulnerability assessment (looking at the exposure of the feature to a pressure), a more detailed scientific vulnerability assessment will be subsequently undertaken to inform the Impact...
Assessments, management measures and enforcement. Defra, MMO, JNCC and NE are currently planning how this will be carried out.

7. Confidence assessments

The sensitivity scores in the sensitivity matrix, and therefore the scores that have been carried through to the tables, have been made through a rapid assessment approach, based on expert judgement. For some features or pressures, there is good knowledge of sensitivity, but for others information is limited. Each sensitivity assessment has an accompanying confidence score. This indicates the relative confidence, according to the criteria (below), indicated by the specialists at the time of making the sensitivity assessment.

<table>
<thead>
<tr>
<th>Confidence score</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Confidence (L)</td>
<td>There is limited or no specific or suitable proxy information on the sensitivity of the feature to the relevant pressure. The assessment is based largely on expert judgement.</td>
</tr>
<tr>
<td>Medium Confidence (M)</td>
<td>There is some specific evidence or good proxy information on the sensitivity of the feature to the relevant pressure.</td>
</tr>
<tr>
<td>High Confidence (H)</td>
<td>There is good information on the sensitivity of the feature to the relevant pressure. The assessment is well supported by the scientific literature.</td>
</tr>
</tbody>
</table>

8. Sensitivity (resistance and recoverability), pressures and pressure benchmarks

Sensitivity: A measure of tolerance (or intolerance) to changes in environmental conditions, made up of:

- Resistance (tolerance/intolerance): response to change, whether an element can absorb disturbance or stress without changing character, and;
- Resilience (recoverability): the ability of a system or feature to recover from disturbance or stress.

Pressure: The mechanism through which an activity has an effect on any part of the ecosystem.

Pressure Benchmarks: The pressure definitions and benchmarks were established by ABPmer and MarLIN under the MB102 sensitivity matrix contract. Where practicable three benchmarks were developed for each pressure, where the benchmarks describe the breakpoints between high/medium and medium/low pressure level, and the mid-point between these two benchmarks (defined as medium pressure). This medium pressure was used for assessing the sensitivity score within the overall sensitivity matrix. To develop the pressure benchmarks, information was drawn from a number of sources including:

- existing benchmarks from other sensitivity assessments (MarLIN website www.marlin.ac.uk);
- environmental quality standards, such as water quality standards established under the EC Water Framework Directive (2000/60/EC);
- guideline values for concentrations of contaminants in sediment and biota (e.g. OSPAR environmental Action Criteria (EAC’s), Canadian Interim Sediment Quality Guidelines (ISQCs);
• initial thresholds developed for indicators of Good Environmental Status under the EC Marine Strategy Framework Directive (2008/56/EC);
• climate change projections (UKCP09);
• expert knowledge of the nature and scale of hydrological changes associated with marine infrastructure developments in UK waters

The pressure benchmarks were further refined following review during the workshops. More information on the development and rationale for the pressure benchmarks can be found in the MB0102 Sensitivity Matrix Report.
Annex 2G – Rock and Biogenic Reef Habitats – Additional detail

Below is supporting information on the targets and indicators which have been proposed by HBDSEG for rock and biogenic reef habitats. It should be read in conjunction with sections 3.5 and 3.8 on advice for selecting indicators and setting targets and covers the three categories of development for indicators (operational now, operational by 2014 (defined by 2012) and operational by 2018).

Existing European Targets and Indicators

*Intertidal species composition & abundance (WFD rocky shore macroalgal tool)* Assessments are undertaken at the water body scale (10-25 km stretches of coastline: www.wfduk.org) throughout the UK and Republic of Ireland. The WFD Macroalgal blooming tool (MAB) is not relevant to rocky shores because nuisance opportunist species only occur on sedimentary shores. There are currently no sublittoral macroalgal or rocky invertebrate community tools used under WFD in the UK. The Rocky Shore Tool Paper v.5 by the Marine Plants Task Team describes the macroalgal indicator in detail (see also Wells et al. 2007).

The rocky shore macroalgal tool is based on the total number of seaweed species found by a defined search procedure on an open coast shore. The tool does not use the abundance of seaweeds because cyclic succession results in large natural changes in seaweed cover in just a few years and this has nothing to do with changes in quality. Correspondingly, seaweed species richness is little affected by total cover and remains constant in the absence of environmental change. However, subhabitat diversity on a shore does affect species richness so differences between shores are taken into account in the assessment.

Although detailed species composition is not used in the WFD macroalgal tool, aspects of the breakdown of the community are used as supporting measures because the % green algae increases with lower quality and % red algae increases with higher quality. Species lists are obtained on single occasion visits to a shore over two hours between May to September and uses a reduced species list (RSL) of 70 species (slightly different lists for different parts of British Isles) to allow for the skill-base of non specialist surveyors. The number of species present is a surrogate for total species. Under WFD species numbers and composition on a shore are translated onto a sliding scale from 0 to 1.0 and organized into five quality categories within this range this is the Ecological Quality Ratio. The relationship between species total and physical shore features is accounted for in this process.

For England and Wales the surveys using the macroalgal tool are done by the Environment Agency (often Wells Marine Surveys, under contract to the EA). The Scottish Environmental Protection Agency have maintained seaweed monitoring efforts but this year voluntary severance schemes have reduced the necessary skill base. The Northern Ireland Environment Agency (NIEA) have detailed coverage of Northern Ireland shores. Although not directly relevant to UK MSFD work, the Environmental Protection Agency in the Republic of Ireland has been fully engaged in the development and implementation of this tool and Norway has since adopted it. Spain and Portugal currently use differing approaches.

Existing UK Targets and Indicators

*Intertidal community indicator (MarClim)*

The intertidal community monitoring undertaken within the MarClim programme is not directly underpinned by a statutory requirement such as WFD. Its development and implementation has been
funded by DEFRA and the UK Conservation Agencies as a climate indicator and has since been used as context to Habitats Directive and Charting Progress national assessments. Like the WFD macroalgal tool, the MarClim monitoring could be double-badged under MSFD to provide wider coverage of Descriptor 1 Indicator Class 1.6.1, i.e. the condition of the typical species and communities. The MarClim programme complements the WFD macroalgal tool because it offers wider taxonomic coverage (covering invertebrate species too), enhanced geographic coverage and therefore confidence in assessment and also offers explicit climate change calibration of assessments (an MSFD requirement). MarClim nevertheless would need some development to meet these new roles. Additional species and habitat types would need to be added to surveys. This should include the incorporation of functional groups in surveys that indicate boulder turning, one of the greatest human impacts on rock shores.

The MarClim method uses a rocky shore species list with a reduced list of temperature sensitive species of invertebrates and macroalage of northern coldwater (N), southern warmwater (S) and non-native (NNS) origins, with range limits occurring in, or near the UK. Baseline data used to support the derivation of the list and selection of monitoring sites is taken from extensive studies in the 1950s, 1980s, 1990s and annual surveys carried out from 2002-date by the Marine Biological Association of the UK around England, Wales and Scotland (Mieszkowska et al. 2005, Mieszkowska 2010, Mieszkowska 2011). The data primarily allows for shifts in geographic distribution and range of individual indicator species to be tracked but also provides a measure of changes in community composition and detection of changes in the dominance of key structural and functional groups such as grazers, space occupiers, predators and primary producers (Mieszkowska 2010).

Categorical abundance data (SACFOR) is collected for the species on the list at each site (including records of absence, recorded as Not Seen). Quadrat counts are also undertaken for N, S, NSS barnacles, N&S limpets and timed searches for S topshells. These data provide information on population dynamics, recruitment success and competitive dominance between N&S species (Mieszkowska et al. 2006; 2007; Poloczanska et al. 2008).

Changes in SACFOR category for individual species will indicate functionally important community level changes through a combination of improvements (increase in category), no change (same category score) or a reduction in condition (1-2 category reduction = ‘deviation’, 3 category reduction = unacceptable, a decline from Common or greater to Occasional, Rare or Not Seen = ‘destroyed’). CCW have methods to assess the scale of bolder disturbance in N2K sites. This methodology can easily be added to MarClim surveys with taxonomic abilities no greater than existing surveys. The location and condition of under-boulder fauna and the presence and location of an anoxic zone on the surface of the boulder can be used to score condition in one of four categories using a systematic sampling strategy (Moore et al. 2009).

**New indicators defined by 2012, operational by 2014**

**Area of subtidal biogenic structures**

Area measures are applicable to structures formed by *Sabellaria spinulosa* (Ross worm), *Sabellaria alveolata* (Honeycomb worm), *Serpula vermicularis* (tubeworm), *Mytilus edulis* (blue mussel), *Modiolus modiolus* (horse mussel), *Limaria hians* (gaping file shell) and *Lophelia pertusa* (cold-water coral) and maerl beds. It is possible that *Ostrea edulis* (native oyster) may also have fallen into this category but any structures formed by this species are probably extinct in UK waters (see Beck et al. 2011).

Biogenic structures occur in a range of environmental settings and possess varying biophysical properties, therefore extent has been measured using a number of methods such as towed and drop-
down video transects, intertidal grid and transects, systematic grab sampling and hydroacoustic survey (eg Roberts et al. 2004; Lindenbaum et al. 2008; Moore 2009; Moore et al. 2009; Stillman et al. 2010).

Extant data have been collected for intertidal fisheries management, assessments of bird food availability for SPAs under the Birds Directive (eg Moore 2009; Stillman et al. 2010), EIAs for developments, and SAC management and monitoring (Lindenbaum et al. 2008; Moore et al. 2009). Methods are generally cost effective and easy to use. However, application is patchy within the UK. Data also needs to be collated and mobilised across agencies and sectors to enable a national MSFD indicator to be assessed.

Repeatability of methods and detection and heterogeneity of the structures influence choice of methods and scale of measurement errors. Differing measurement errors will need accommodation during combined assessments: scale-up to appropriate assessment units may be necessary. Extant monitoring needs systematic collection and collation and geographical expansion of extant schemes to get appropriate coverage.

**Area of Subtidal rock**

Widespread active monitoring for this indicator is also not cost effective but collation of extant data to underpin a desk-based assessment would be feasible.

Extent measures at UK or regional sea level for subtidal rock are available from a combination of models and multibeam data (ref MESH & Robinson et al.). Direct monitoring of subtidal rock at a UK scale is not cost effective. A desk assessment can be undertaken for this indicator as it was for Charting Progress 2 (DEFRA 2010). Up to date rock extent maps would need to be collated and overlaid with pressure data to simultaneously measure and assess extent and the area over which community change may have occurred due to anthropogenic pressure.

**Intertidal rock extent (inc exposure sub-types)**

Burrows (submitted) and other have modeled and ground truthed shore types and thereby estimated the extent of different exposure types and the relative proportions affected by coastal developments. Some areas have ground-truthed Phase 1 intertidal survey data (Wyn et al. 2000) enabling area measures of shore types but for Scotland and elsewhere high coastal complexity and scale make linear extent the achievable option and common denominator at a UK scale. This indicator is cost effective, easily measured, and achievable as a desk-based study using ground-truthed model data and development information.

**Area of littoral chalk habitat**

Most areas of intertidal chalk in the SE of England have been mapped (Natural England and Environment Agency) but other sources of information may be required from elsewhere. Some new data will be required but this indicator can be achieved with systematic data collation efforts. Chalk habitats are nationally rare and have been historically lost during coastal development and defense. The area or linear extent of this habitat can be systematically assessed using a desk-based approach, collating data from within and outwith SACs. Natural Englands (NE) Casework Tracker database has suitable data on coastal development and mapping and other data in are available in SE England NE, Wildlife Trusts, Shore Search, Marclim and E. Sussex CC.

**Area of intertidal seacaves**

This potential indicator would need to be a desk-based assessment of known developments; a baseline of all sea caves is not achievable.
It is uneconomic to actively monitor sea caves at a UK scale because some parts of the UK are highly complex with many sea caves and most are unaffected by anthropogenic activity. In some coastal regions, however, they have been bricked-up in coastal development work and in these areas ad hoc monitoring does occur in response to coastal development pressure.

Abundance of associated species on biogenic reef
Community-based indicators are applicable to several types of biogenic structures: Sabellaria alveolata, Serpula vermicularis, Mytilus edulis, Modiolus modiolus Limaria hians and Lophelia pertusa. Data are available from several types of biogenic structure in the UK (e.g. Rees et al., 2008; Sanderson et al. 2008; Trigg et al., 2011) using directed sampling by divers, towed video and intertidal coring. Methods are cost effective and easily applied but small-scale heterogeneity requires careful consideration of deployment strategy, stratification and statistical power in most cases.

Community composition varies between biogenic structures of the same species so this indicator and it’s targets will need to be derived from assessments on a site by site basis in the first instance. Physical impact models need incorporation e.g. those for Modiolus, Serpula and Limaria (Cook et al. in prep, Moore et al. 2009, Service & Magorrian 1997). Application of multivariate and univariate indices (inc WFD multimetrics) needs evaluation at the scale of a UK MSFD indicator. Wider geographic coverage will be required than present sporadic monitoring in UK SACs. Community based indicators for Sabellaria spinulosa will require careful consideration.

Density of biogenic reef forming species
This indicator is potentially applicable to several types of biogenic structures: Sabellaria spinulosa, Sabellaria alveolata, Mytilus edulis, Modiolus modiolus, Limaria hians and Lophelia pertusa. In common with the preceding indicator, an understanding of appropriate targets needs to be constructed within a model of state that is yet poorly understood for many of the biogenic types. Wider geographic coverage will also be required despite various local monitoring activities in SACs (e.g. Rees et al. 2008; Sanderson et al. 2008). Methods are cost effective and easily applied but small-scale heterogeneity requires careful consideration of deployment strategy, stratification and statistical power in most cases. Lophelia reefs in NW Scotland may present substantial experimental and logistical hurdles.

Epifaunal indicator species
Widespread drop camera work would be needed to make this indicator operational and there may be scope for new towed technology application. The abundance per unit of area of erect taxa can be determined for a unit of video footage. Many of these taxa are slow-growing, sessile and vulnerable to physical abrasion although a pressure gradient model may be need to be tested to determine target levels.

This indicator is closely related to the Subtidal species composition & abundance (sponge / anthozoan) indicator above but addresses horizontal rocky habitats, generally in deeper water, and over a wider area. Sufficient monitoring in deeper habitats would be relatively more expensive but there are efficiencies to be gained by using the same platform as for other indicators.
New indicators operational by 2018

**Subtidal species composition & abundance (sponge / anthozoan communities)**
Fragile sponge and anthozoan communities on subtidal rocky habitats have been studied using divers in circalittoral steeply inclined rocky habitats where erect sponges and anthozoans dominate in some UK protected areas. They have also been studied in more horizontal and often deeper habitat settings using drop-down and towed video (Erect epifaunal indicator species - below). A national indicator would require a dual approach to both broad types and stratification to particular biotope types in each case. A number of sentinel monitoring stations would be required. This indicator would be indicative of wider circalittoral communities and potentially sensitive to abrasion. Although cost effective and easily measured, the establishment of monitoring stations and supporting case studies would require some investment.

**Sponge diversity**
Sponge dominated communities occur widely in the UK shallow circalittoral. Morphological richness and diversity measures in sponge communities are a useful, cost effective surrogate for sponge species richness and diversity (Bell & Barnes 2001; Bell 2007) and there is evidence elsewhere in the world that sponge species diversity is responsive to water quality. Morphological monitoring baselines for some sponge communities have been developed in Welsh MPAs but the ecological response model remains untested within Atlantic Europe. Developing this indicator would require extended geographic coverage and a test of the community model response.

**Kelp depth and kelp park depth**
Kelp depth is linked to light attenuation (eg Kain 1979; Dayton 1985). In recent trials the max depth bcd at which kelp and kelp park occurs can be precisely measured and data are currently available for some parts of the UK. Historic data may also be retrievable. Burrows (submitted) shows a link between biodiversity and kelp depth, therefore a measure of kelp depth can be used as a cost effective surrogate for infralittoral biodiversity (but will need periodic direct measurement). A national case study would verify the infralittoral biodiversity- kelp depth link and serve as a repeatable reference point for periodic re-survey. Monitoring site selection needs to include appropriate geology. Some turbid regions such as SE England may not be appropriate. Remote sensing data on turbidity may offer additional context for this indicator.
Annex 2H – Sediment Habitats – Additional detail

Below is supporting information on the targets and indicators which have been proposed by HBDSEG for sediment habitats. It should be read in conjunction with section 3.6 on advice for selecting indicators and setting targets and section 3.7 on advice on setting baselines. All sediment indicators which are not already in use within the WFD fall into the category of ‘operational by 2014, defined by 2012’ and therefore this is the only category considered here.

The sediment habitat list consists of 12 predominant habitats and 16 special habitats. Other habitats originally listed under sediments were considered to be more appropriate to be dealt with as rock and biogenic reef habitats, namely, Coral gardens, Ostrea edulis beds and Deep sea sponge aggregations. The only sediment habitat listed not occurring in UK waters (North Sea and Celtic Sea sub-regions) is Cymodocea meadows. The proposed sediment indicators in this report are suggested in most cases as being applicable to all sediment habitats although the value of using indicators such as range and extent for some widely distributed predominant habitats such as shelf and abyssal sediments is probably limited. The Redox Potential Discontinuity (RPD) / Sediment Profile Imagery (SPI) indicator suggested for development will have limited applicability where sediments are coarse/highly mobile due to the lack of any RPD layers and difficulties in utilising SPI.

Other indicators are specific to habitats such as the WFD seagrass (Zostera beds) and the Opportunistic Macroalgae tools (intertidal mudflats and possibly littoral sand). As with the seagrass tool, the Infaunal Quality Index (IQI) has been developed for coastal and transitional waters under WFD so will require further work to apply in offshore environments. The WFD intertidal seagrass tool has been in development since 2004 and entered use operationally in 2007. It has been developed and tested at individual beds and water bodies in different European countries. The UK and Republic of Ireland Marine Plants Expert group have agreed a common matrix for allocating status to intertidal seagrass assessments. The benthic invertebrate soft sediment IQI tool has a long history of research into its use as an index for assessing ecological status of benthic invertebrate communities. The intertidal opportunistic macroalgae tool was developed from 2003 and has been used operationally for WFD since 2006 (components of the opportunistic macroalgae tool have been used for assessment for Urban Waste Water Treatment Directive since the late 1990s). The IQI and opportunistic macroalgae tools were used for reporting ecological status in the first round of WFD River Basin Management Plans. The seagrass tool, IQI and opportunistic macroalgae tools have also been validated against comparable assessment methods of other member states bordering the North East Atlantic through the WFD Intercalibration process. A saltmarsh tool is under development (Best et al 2007) but does not have the same kind of evidence base as exists for the other tools at present although there have been England and Wales-wide assessments of extent and in many waterbodies. This provides a useful baseline against which repeat surveys can be made and the saltmarsh tool refined for broader use.

For the WFD intertidal seagrass indicator there is already some monitoring in place for WFD whereby bed extent is assessed by directly tracking around the bed or remotely through aerial imagery and quadrats placed to obtain % cover. The WFD-UKTAG report also suggests that the extent of seagrass beds may in some cases be measured by remote imagery. Sampling programmes have been established for IQI in a limited number of water bodies under WFD but there is currently no offshore monitoring in place for IQI. Sampling occurs for the opportunistic macroalgal tool with sites chosen by stratified random sampling within the intertidal zone. Saltmarsh assessments will be based on quadrat sampling along stratified random transects to examine changes in vertical zonation within the intertidal zone as well as aerial extent using aerial imagery. The saltmarsh tool is expected to be used operationally in 2012. Although indicators will mostly be applicable across habitats, targets will vary according to the habitat under consideration. The expert advisory group on sediments also suggests
that other habitats may need to be identified under the ‘widespread habitats’ category and more urgently under the category ‘habitats which merit a particular reference’.

**New indicators defined by 2012, operational by 2014**

**Distributional Range of Habitat (Indicator 1.4.1)**

This indicator is suitable for establishing the geographical range of a habitat (i.e. northern and southernmost limit National Grid Reference (NGR), lat/long), both at a large scale (e.g. UK) and a smaller scale (e.g. within a region sea). It would be more useful for habitats that are at risk of a retraction in range (e.g. saltmarsh) rather than those for which long-term changes are unlikely (e.g. Abyssal sediment).

**Distributional pattern of habitat (Indicator 1.4.2)**

This indicator is tightly linked to the ‘range of habitat’ indicator but would show distribution information where that is thought to be important. For example, it may be useful to see distance between Zostera (seagrass) beds or other special habitats as this can be linked to connectivity of systems and the ability of habitats to be maintained through dispersal of larvae/propagules from other populations.

**Area of sediment habitat (Indicator 1.5.1)**

This indicator would look at the spatial extent (area) of all non-intertidal sediment habitats (predominant and special) establishing the location (NGR, lat/long) and boundary of habitat (NGR, lat/long). The reason ‘intertidal habitat area’ and ‘habitat area’ are proposed as two separate indicators is that there is a lot of information on the location and distribution of intertidal sediment habitats whereas many of the subtidal sediment habitats will have to rely on mainly modelled maps. There is also a difference in pressures that the intertidal and subtidal regions are subjected to and there is a greater array of management and policy drivers at the terrestrial/marine boundary. One of the most important issues in utilising this indicator will be to separate changes in habitat distribution/area caused by anthropogenic impacts from changes due to new information becoming available from surveys. For example, a 5% loss in a habitat could go unnoticed if the equivalent amount of habitat is ‘found’ due to improvements in modelled distribution based on new survey data. There is an additional issue around our ability to measure extent at present as there are currently significant errors associated with the modelled approach and even techniques used in the field which have their own associated errors (e.g. position-fixing and instrumentation error). Expert judgement and models will continue to play a significant part in this judgement and have done so to date for Habitats Directive.

**Redox Potential Discontinuity / Sediment Profile Imaging (Indicator 1.6.3 and criterion 6.2):**

It is proposed that an indicator be used based on sediment profile imaging (SPI) with sampling taking place on a decadal basis for offshore environments (but could be sampled more frequently in coastal areas where some current monitoring already exists). The preferable approach would be to use SPI to just measure benthic habitat quality index (BHQ), an approach described by Rosenberg et al (2009) who related it to EU-WFD environmental quality status. The indicator could therefore be applied to a wide range of coastal and offshore habitats including deep sea sediments (Diaz and Trefry 2006) although it would need some refining for deep sea areas. It would also need to be supplemented with some conventional quantitative sampling.

Sediment Profile Imaging (SPI) has been used for many years as a pollution monitoring technique by evaluating the activity of resident marine fauna (O’Reilly et al 2006; Keegan et al 2001). This means that for certain areas (e.g. Galway Bay and Kinsale Harbour) there are now several years of SPI data (P. Dando – pers. Comm.) and other UK laboratories are investigating SPI as a tool for long-term
benthic monitoring. Challenges such as removing the subjective nature of interpreting results have been addressed by developing specialist software (Geeta et al 2004). Recently there has been more interest in utilising SPI techniques to provide indicators for WFD and MSFD. For example Birchenough et al (2011) look at using two metrics, bioturbation potential (Bpc) calculated from quantitative information and apparent Redox Discontinuity Layer (aRPD) derived from SPI images as an indicator tool to assess seabed structure and function of ecosystems. As Bioturbation Potential (Bpc) is very costly to calculate for different sites and cannot be derived from SPI data (P. Dando – pers. Comm.) the preferable approach would be to use SPI to just measure benthic habitat quality index (BHQ), an approach described by Rosenberg et al (2009) who related it to EU-WFD environmental quality status. The indicator could therefore be applied to a wide range of coastal and offshore habitats including deep sea sediments although it would need some refining for deep sea areas. There may also be some difficulty with coarser sediments and even in the deep sea penetration depth may be an issue. There would also need to be ‘ground-truthing’ using a box corer or similar deep sampling device (so it samples at an adequate depth). This would also be undertaken alongside the SPI based monitoring at a decadal scale.

Additional supporting information on advice on setting baselines for sediment habitats

For the WFD seagrass, opportunistic macroalgal and saltmarsh tools, reference conditions are derived using a combination of historic data and expert judgement. For the Infaunal Quality Index (IQI) reference conditions are derived using a combination of best available low pressure data and expert judgement with reference conditions being adapted according to habitat. Reference conditions for the IQI are under continued development as data becomes available.

For the indicators ‘range of Habitat’, ‘spatial distribution of habitat’, ‘intertidal Habitat area’ and ‘habitat Area’ it is important to consider baselines (and targets) established under other directives and policy reports for these quantity elements i.e. Favourable Conservation Status (Habitats Directive), Good Ecological Status (Water Framework Directive), thresholds for ‘threatened and declining habitats and species’ (OSPAR) and ‘area impact assessments’ (Charting Progress 2 Habitats assessment). For a baseline, Charting Progress uses historical conditions i.e. a concept of ‘undisturbed conditions’. OSPAR also uses a historical baseline and Good ecological status under the Water Framework Directive is equivalent to undisturbed conditions.

The Habitats Directive takes a slightly different approach in using a baseline which incorporates the concept of viable area of habitat against which to assess habitat loss. The setting of this baseline can be current conditions (if the area is considered to be ‘viable’) but can also use historical data to construct a viable area, if required. Nine of the special sediment habitats are covered by baselines and targets as part of the Habitats Directive, with the remaining five special habitats being covered by OSPAR. The twelve predominant habitats were all assessed as part of Charting Progress 2 with baselines and targets based on a combination of OSPAR and Habitats Directive thresholds. For GES, it is proposed to retain all the baselines as set out in these policy drivers whilst recognising the challenges of providing historical baselines for any habitat. It is also important to note that for at least two of the habitats (Atlantic salt meadows - Glauco-Puccinellietalia maritimae’ and ‘Zostera beds’), information on habitat extent is included as part of the assessment of GECs for WFD).

For the pressure related condition indicators ‘distribution of pressures’ and ‘percentage seabead adversely affected by human activities’ the baseline would be the based on the assessment undertaken for Charting Progress 2 (and any available updated information). As already stated, ‘changes beyond prevailing conditions’ is not an indicator in itself with an associated baseline and target but really forms part of the context for all of the condition indicators, to allow anthropogenic

1 http://www.oceanlab.abdn.ac.uk/research/spi.php
changes to be identified. The information from the monitoring of changes in prevailing conditions, along with information on ocean processes would allow an understanding of current baselines of state for sediment habitats.

For the SPI indicator the baseline will have to be set using expert judgement making sure that sampling was undertaken in a way that took into account various factors such as seasonal variation in RPD depth.
ANNEX 3 Pelagic habitats report

The Development of UK Pelagic (Plankton) Indicators and Targets for the MSFD

This annex is meant to supplement the information contained in the actual report with technical detail. Repeating information and content has been omitted from the annex in the interest of brevity.


1. Lifeforms and State-Space theory

1.1 Introduction
A lifeform is a group of species (not necessarily taxonomically related) that carry out the same important functional role in the marine ecosystem. For example, diatoms as a group of species have a functional role related to silicon cycling. In this report, we are concerned with methods for quantifying the state or health of one part of marine ecosystems, the pelagic community of organisms, also called the plankton. The essential features of the method proposed are (i) the grouping of the many species of organisms found in the water column into a few lifeforms, and (ii) the display of changes in the abundance of each of these lifeforms using a state-space approach.

In the main report we provide the full set of lifeforms that we propose to use to develop indices for the pelagic habitat component of the biodiversity group of descriptors plus Descriptor 5: Eutrophication. At this point the theory underlying the method is described with reference to an example pair of these lifeforms described here. The phytoplankton includes many species of microscopic algae. Some of these species belong to a group called diatoms, which are characterized by having a cell wall made of silicon. Diatoms are tolerant of the turbulent, low-light, conditions of spring in temperate seas, and so, characteristically, give rise to a spring phytoplankton bloom in March, April or May. Many planktonic animals and fish lay their eggs to hatch in time for this bloom, which provides food for the growth of the larvae. Dinoflagellates comprise another taxonomically-defined group within the phytoplankton. They do not require silicon, typically become abundant only after the spring diatom bloom, and are characterized by two flagella (whiplash like attachments) which enable them to swim up or down and so take advantage of water-column layering, for example in estuaries or when summer warms the surface layer of the open sea. Some provide a source of food for planktonic animals during the summer, but some species contain toxins that may deter grazing and are poisonous to some animals and humans.

1.2 The basis of the state-space approach
The theory of state-space derives from physics and especially the discipline of thermodynamics, but has been adapted to apply to systems in general and in our case to ecosystems. A system is defined as ‘a set of components and relationships within a defined boundary’ (Tett et al. 2011). To identify the state of a system it is necessary to define a set of system state variables. These are attributes of the system that change with time in response to each other and external conditions. There needs to be enough variables in the set to jointly describe all system variability other than that (perhaps somewhat arbitrarily) defined as ‘noise’.
Consider that our aim is to ascertain the state of the phytoplankton in a defined part of the sea, on a given day. A representative water sample is collected and examined microscopically to obtain a list of the species present and their abundances. While mapping of data would be preferable in some cases (D1.4: Habitat distribution) and can indeed be carried out in open waters thanks to the Continuous Plankton Recorder, this is not possible with time-series that collect data at just one or two or twenty stations. However, we could see a change in taxa abundance that becomes increasingly apparent northwards or southwards which may signal a shift in distribution. Therefore we use abundance as it is a metric that is routinely recorded by monitoring programmes and that is applicable to the relevant MSFD criterion. The total abundance of all the diatoms and the total abundance of all the dinoflagellates are calculated giving two numbers, which give the co-ordinates of a point that can be plotted into a space, or map, defined by two axes: one for the abundance of diatoms, and the other, at right-angles, for the abundance of dinoflagellates (Fig. 1).

Figure 1. An illustration of how the state of a diatom-dinoflagellate community is defined by a point plotted into state-space.

This point represents ecosystem state at the instant that the water sample was taken and as characterised by the abundances of diatoms and dinoflagellates. However, as noted earlier, a characteristic of the phytoplankton is its high natural variability (on temporal scales that range from days to inter-annual). It is likely therefore, that analysis of a water sample taken a few days or weeks later from the same location might give abundances that plotted to a different point in the diatom-dinoflagellate state-space. The path between the two states is called a trajectory, and the condition of the phytoplankton is defined by the trajectory drawn in the state-space by a set of points. The seasonal succession of species in seasonally stratifying seas means that this trajectory tends in a certain direction as dinoflagellate abundance increases relative to diatom abundance during summer. However, as phytoplankton growth declines during autumn, abundances decrease towards levels prior to the spring bloom, with the result that the trajectory tends towards its starting point (Fig 2). Given roughly constant external pressures, the data collected from a particular location in the sea over a period of years forms a cloud of points in state-space that can be referred to as a regime.
As Fig. 2 shows, our argument is that some changes in external conditions - for example the consequences of an oil spill - might cause a temporary deviation from the usual regime - while other changes - for an example, an increase in the inputs of human generated nutrients - might cause a permanent deviation from this regime, by causing the system to switch to a new regime. The Plankton Index (PI) tool, described later in this section, provides a way to quantify movements in state-space away from the ‘usual regime’.

![Figure 2. Regime shift in state-space (Tett et al., 2007).](image)

It is unlikely that two state variables will be sufficient to describe all important variability in the marine pelagic system. In principle this is not a problem: we simply add other axes to the state-space map. This is illustrated for 3 phytoplankton state variables in Fig. 3. Note that the third axis has to be drawn at right-angles to the other two, and the Figure is in fact a 2-D projection of a 3-D object. The rule is that each additional state variable has to be independent of the existing set, and its axis has to be drawn at right-angles to all existing axes. In principle, therefore, the state-space map has to be drawn in as many dimensions as there are state variables but this is difficult. Our solution is to rely on sets of state-space diagrams, each in two dimensions. As long as each axis in any plot is independent of all other axes in any plot, and we follow the rule that all axes must be commensurable it will be straightforward to combine results from any number of plots into a single Plankton Index.

We refer to these state-space diagrams as ‘maps’, and to the lines that link points as ‘trajectories’ rather than ‘graphs’. In normal scientific usage, a graph implies a functional relationship between the values on the horizontal (x-axis) and the values on the vertical (y-axis). That is, a change in x causes a change in y. In the case of state-space diagrams, there is no implication that change in one state variable causes change in another, although change in both might be linked in some way. In the diatom-dinoflagellate example, although both lifeforms compete for supplies of nutrients and energy, there is no direct, functional link that allows us to say that diatom change causes dinoflagellate change. Just as, in the case of a map of the Earth’s surface, it makes no sense to say that changes in latitude cause changes in longitude: instead, latitude and longitude are the two co-ordinates that define a position. Thus, when referring in a general way to the two axes of a state-space plot, we label them as ‘Y1’ and ‘Y2’ in contrast to the ‘X-Y’ labels used in a graph that implies a functional relationship.
1.3. Why state-space?

We could of course simply plot the two example time-series of diatom and dinoflagellate abundances as graphs against time (Figure 4), adding additional data as required. We could simplify the picture to some extent by, for example, plotting a time-series of the ratio of dinoflagellates to diatoms, or the percentage of total phytoplankton abundance contributed by diatoms. And then we could extract simple statistics, such as annual means of diatom abundance or percentage diatoms. But such a method throws away information about the annual succession of lifeforms, exemplified by seasonal changes in the relationship between diatoms and dinoflagellates, which seems an important aspect of the pelagic ecosystem. Indeed, this can be seen as forming part of the structure of the ecosystem, substituting in temporal variation in terms of spatial structure. Furthermore, any index based on an annual statistic is sensitive to sample collection routines: it is, for example, easy to miss the spring phytoplankton bloom.

These objections were recognized during the development of indicators for the phytoplankton biological quality element of the Water Framework Directive, and methods developed for the construction of seasonal envelopes of variation for each lifeform. Nevertheless, an approach based on state-space, although initially appearing complex, has several advantages. The first is that of potential conceptual consistency across the variety of animals and micro-organisms that contribute to the ecological status of the pelagic community. The second is that this consistency leads to a very simple method (that of counting points) for measuring change. The third is that the state-space approach lends itself to simple visualization: experience suggests that most people find pictures (geometry) easier to understand than complicated numbers (algebra).

One objection to state-space as opposed to time-series graphs might be that a state-space plot results in a loss of information, about the time-dependency of changes in abundance. The main justification is that system state is not defined by time but by the instantaneous values of state variables; two systems that have the same pair of values for $Y_1$ and $Y_2$ are said to be in the same
A practical advantage is that state-space plots are less sensitive, than statistics based on time-series graphs, to defects in sampling regimes. Nevertheless, it is important to sample throughout the year so that the plankton regime is fully characterised.

1.4 Estimating a value of a Plankton Index for a pair of lifeforms

The operations necessary to get a value of a Plankton Index for a pair of lifeforms are listed in the complete Belfast meeting report (contact Abigail McQuatters-Gollop abiqua@sahfos.ac.uk for a copy). Many of these operations can be demonstrated using spreadsheets, drawing by hand, and counting by eye. Nevertheless, a previously-written (and debugged) computer program allows easier routine operation. The results were made by a program written with MatlabTM, software that includes an extensive library of mathematical and graphical functions.

The starting point was the time-series of diatom and dinoflagellate biomass at the L4 station in the English Channel near Plymouth. Figure 4 shows graphs of biomass (Y) against time (t), which is to say the position of a point is defined by its Y-t co-ordinates. For example, the co-ordinates of the point for diatom biomass on 2 May 1994 are $Y_1 = 2.53$ and $t = 1994.442$. The corresponding dinoflagellate co-ordinates are: $Y_2 = -0.31$ and $t = 1994.442$. The Y-coordinates are in fact logarithms (to base 10) of estimates of 337.42 diatom and 0.48 dinoflagellate biomass (units). Logarithmic transformations are commonly applied to data on plankton (Barnes, 1952) because they allow more reliable statistical analysis and interpretation, and also allow change at low abundance to be seen as clearly as change at high abundance. In essence, a given amount of change on a logarithmic axis shows the same proportionate increase or decrease, irrespective of abundance. Such a transformation is also desirable because it ensures commensurability of axes in state-space plots.

![Figure 4. Time-series of diatom and dinoflagellate biomass from the Plymouth L4 station, data provided by Claire Widdicombe at PML.](image)

The next step is to make such a state-space plot for which the co-ordinates of each point are a pair of values of $Y_1$ and $Y_2$ (such as 2.53 and 0.48 for L4 on 2 May 1994). A minor difficulty arises when
there is no value of $Y_2$ corresponding to $Y_1$ at the same time (or vice versa), but it is sometimes possible to approximate the missing value.

In order for a Plankton Index (PI) to be calculated, it is necessary to establish reference (baseline) conditions as the basis for subsequent comparison. **The term reference is used here simply to denote the data set against which comparisons will be made (baseline), and does not imply pristine conditions.** In the example, Plymouth L4, the reference (baseline) period was taken as the 4 years from 1992 through 1995, during which sampling had taken place at roughly fortnightly intervals. Plotting the 4 years of data gave the cloud of points shown in the left-hand part of Figure 5.

![Figure 5. Example of the calculation of a component of the Plankton Index. In this case, the diatom-dinoflagellate pair using data from the Plymouth L4 station. Data from 1992-1995 has been used for the reference set, on the left side; the right side shows a comparison of data from 2001 with this envelope.](image)

Next, we want to define a reference (baseline) envelope to include all, or most, of these points, and to give us a basis for comparison with data collected in subsequent years or at other sites. The outer part of the envelope is made by applying a geometric method known as a Convex Hull (Sunday, 2004; Weisstein, 2006) to the cloud of data points plotted in state-space. The outer points can be thought of as pins pushed into the plot, and the Hull as a rubber band stretched around these pins.

In principle, the reference envelope defines a bundle of trajectories, and in some cases, such as phytoplankton, limitation theory suggests that the bundle should have a hollow centre (Tett & Mills, 2009). It is possible to fit an inner envelope by turning the points ‘inside-out’ around the centre of the cloud of points, applying the Convex Hull procedure again, and re-inverting.

Tett et al. (2008) found that the size and shape of the envelope was sensitive to sampling frequency and total numbers of samples. Envelopes were made larger by including extreme ‘outer’ or ‘inner’ points, and the larger the envelope, the less sensitive it was to change in the distribution of points in state-space. Thus, it is desirable to exclude a proportion ($p$) of points, so as to eliminate these extremes and obtain a smaller, tighter, envelope. The envelope, thus drawn, defines a domain in state-space that contains a set of trajectories of the diatom-dinoflagellate component on the marine pelagic ecosystem and thus represents the prevailing regime during the reference period. It is desirable to include 3 years of data in drawing the envelope, in order to take account of natural inter-annual variability; but not too many years (no more than 5), because Plankton Indices are tools to examine change in time.
The next step plots a new set of data into state-space and compares them with the reference (baseline) envelope. Does the new cloud of points fall mainly within this envelope or instead show a shift in state-space? The right-hand side of Figure 5 illustrates this. Experience suggests that fewer new points are needed for the comparison than are desirable for the reference envelope. Currently we think that it is desirable to have at least a dozen points for comparison. These should represent samples taken throughout the year, because seasonal variation is seen as an essential part of the ‘structure’ of the phytoplankton community.

The value of the PI is the proportion of new points that fall inside the envelope, or, to be precise, between the inner and outer envelopes. In the example, for the comparison year 2001 at L4, 22% (or 9) of 41 new points lie outside, and the PI is 0.78. A value of 1.0 would indicate no change, and a value of 0.0 would show complete change, with all new points plotting outside the reference envelope. The envelope was made by excluding 10% of points, so some new points are expected to fall outside: four, in the case of the example. Is 9 significantly more than 4? The exact probability of getting 9, by chance alone when 4 only are expected, can be calculated using a binomial series expansion, or approximately, by a chi-square calculation (with 1 df and a 1-tail test). The conclusion is that the value of 0.78, is significantly less than the expected value of 0.9, and so conditions in respect of diatoms and dinoflagellates in 2001 were statistically significantly different from those in 1992-96.

What is the meaning of this change? It could be the result of no more than ‘normal’ inter-annual variation, which might take the system outside the reference (baseline) envelope without indicating a persistent shift in regime. Thus the next step is to examine a trend. There were sufficient data available for L4 to allow a comparison to be made for individual years, from 1997 to 2002. As plotted in Figure 6, the values of the PI (for the diatom-dinoflagellate state-space component) fluctuate from year to year, with some of the values of the index for particular years showing a significant proportion of new points falling outside the reference envelope. However there is no significant temporal trend in the values of this PI.

![Figure 6](image.png)

**Figure 6. A time-series of the PC-LF a component of the Plankton Index. In this case, the diatom-dinoflagellate pair using data from the Plymouth L4 station. The index for each year is calculated by comparing the state-space plot for each year against the 1992-1995 reference set.**

### 1.5 Composite Plankton Indices

A composite Plankton Index is put together from several PIs made as described above, each involving a 2-D state-space plot. To avoid ambiguity, we will notate a component value as $PI_i[t, t_{ref}]$, referring to component i (e.g. that for diatoms and dinoflagellates shown above) for
year \( t \) compared with the reference period \( t_{\text{ref}} \). The overall Plankton Index for a given year is simply the mean of the available component PIs, or:

\[
PI_t = \frac{1}{n} \sum_{i=1}^{n} PI_{i,t_{\text{ref}}}
\]

(To repeat a prior stipulation, this procedure requires all axes to be commensurable (Box 4.6 in the Belfast report) and no lifeform to appear more than once in the set of axes used for the overall analysis.) We can assess the significance of a value of \( PI_t \) using the same approach as for assessing the significance of the diatom-dinoflagellate PI component, i.e. by summing the total number of points that fall outside all component envelopes and comparing with expectation based on the proportion excluded from the reference (baseline) envelope. As in the case of the diatom-dinoflagellate example, a time-series of values of the compound PI can be examined for trend. What is to be done if a trend is found, will be considered in the next chapter.

Such a composite PI might include components for phytoplankton, heterotrophic microplankton, and zooplankton. We contend that this would provide a single holistic indicator of changes in the condition of the pelagic ecosystem. In addition, lesser compilations can be made, to provide indices relevant to particular MSFD descriptors. If we reserve the label PI for the holistic indicator, we could refer for example to an eutrophication-relevant PI as \( PI(D5) \), and write \( PI(D5)[t = 1990:2010] \) for the time-series of values covering the comparison years 1990 through 2010.

1.6 Discussion

To recapitulate: we have argued that the plotting, in state-space, of values of the abundance of several lifeforms belonging to the plankton, enables the tracking of changes in the condition of the pelagic community over time, by means of comparing the state-space positions of new points with a reference envelope. As mentioned above, the use of the term ‘reference(baseline) envelope’ is not meant to imply that the conditions it described are pristine, or correspond to ‘reference conditions’ as the term is used by the WFD, or, necessarily, to GES as used by the MSFD. Our method is aimed at providing a tool for management, able to show whether condition is changing. If it is changing, then time-series of PIs can be examined for possible correlations (in space or time) with time-series of pressure indicators.

However, the MSFD explicitly requires the establishment or maintenance of GES over marine sub-regions, and so it will be useful to establish reference envelopes for ecohydrodynamic water types in which the pelagic environment is deemed to be of good status. Establishing reference conditions for the WFD has proved troublesome, because it has involved finding water-bodies subject to no, or very little, anthropogenic pressure. Because the MSFD defines GES in terms of proper ecosystem functioning, it would seem possible to find homogenous water bodies (i.e. spatial regions of constant ecohydrodynamics, within MSFD sub-regions) where ecosystem functioning can be explored through a combination of detailed investigation and numerical modelling, and thus establish GES reference envelopes for the PI tool. In the medium term it might be possible to make estimates of such envelopes using ecosystem models alone.
2. Meeting targets
2.1 The assessment process

For the pelagic habitat component of the biodiversity group of descriptors, the criteria and indicator target are the same and as are applicable to changes in floristic composition under Descriptor 5, eutrophication. The target is:

*The plankton community is not significantly influenced by anthropogenic pressures.*

Assessment scales are discussed in the main body of the report but are reiterated here. To assess the environmental status of the plankton at the regional sea level, it is important that sampling stations are located in each ecohydrodynamic region in UK waters. Ecohydrodynamic regions are bodies of water that are distinct from each other as a result of stratification (vertical layering of water masses) or differences in mixing. Such a regional spread of data is necessary in order to ensure that assessment is spatially representative. Figure 7 shows the simulated distribution of ecohydrodynamic regions (permanently stratified, permanently mixed, intermediate regions and regions of freshwater influence - ROFI) in the North Sea based on the 3D General Estuarine Transport Model (GETM) physical model (Cefas unpublished data). When monitoring station are spatially isolated but have a time-series (such as the numbered stations in Figure 7) they can be assessed independently for significant changes. The open sea regions, and coastal areas with WFD monitoring, can be aggregated for assessment based on the ecohydrodynamic breakdown.

A decision tree showing how the lifeform indices and Plankton Index will be used to determine whether the target has been met at the scale of assessment (ie open sea ecohydrodynamic regions, coastal zone ecohydrodynamic regions if data allow, individual monitoring stations) and GES of the plankton confirmed is outlined in Fig. 8. For each individual monitoring station, annual estimates (based on comparing data from each year with the baseline data) of the Plankton Index for each descriptor (PID1, PID4 and PID5) and the overall Plankton Index (PI) will be used to construct time-series. The high natural variability in the plankton is likely to cause some inter-annual variability in values of each index and on occasions differences between years may be statistically significant. Figure 9 illustrates this point. The figure shows the diatom and dinoflagellate state-space plots (data from the Plymouth Marine Laboratory L4 time-series) for the reference years 1992-1995 and data from 2001. The value of the index was statistically significantly different indicating that the diatom and dinoflagellate community in 2001 had changed compared to the reference years. The test to determine whether a time-series is showing a change in the plankton rather than natural variation will be the presence of a statistically significant trend or change. An example of a time-series for the PCI-LF component of a PI based on the Plymouth Marine Laboratory L4 time-series data is shown in Fig. 9. In this case there is no significant change in the index.

The absence of a significant change will show that there has not been an overall change in an index. A significant change in an index will be attributed to human pressure (and not climate change) if there is a significant correlation between the change in an index and a particular human pressure and if that change is not seen at other coastal stations or in the open sea. The absence of a significant correlation will be used as evidence that the target for Good Environmental Status of the pelagic community as a whole, and the pelagic (plankton) components for the biodiversity, food-web, seafloor integrity and eutrophication (D5.2.4) descriptors has been met. Identification of a significant correlation to a pressure will signify that the target has not been met and this may require investigative monitoring and a programme of measures.

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2 This presupposes that the starting point of the trend (baseline or reference conditions) represent GES.
Figure 7 The simulated distribution of a 50 year average of ecohydrodynamic regions (with monitoring stations overlain) in the North Sea. Based on a 3-D General Estuarine Transport Model (GETM) physical model (Cefas unpublished data). Areas in white have no consistent dominant level of mixing or stratification.

One way of visualising whether the target has been met on a regional and local scale is by colouring time-series plots. Figure 9 shows an example in which, the green background graphs indicate no trend. A red background signifies a significant trend in the index that might be up or down. In this example there are only trends in near-shore waters, as the offshore waters show no trend suggesting a localised pressure rather than a broad scale climatic effect. In both Liverpool Bay and the Wash LF group 1 are red (suggestive of a common pressure e.g. nutrients), but in the Wash/Thames area LF group 4 is red also suggesting an additional pressure. However, there is one important issue that is relevant to all of the MSFD descriptors: how to determine whether the assessment region as a whole is in Good Environmental Status if the target is not met in one location. Similarly, can the environmental status of an assessment region be deemed to be good if the pelagic habitat targets have been met for the region as a whole, but the target for a different ecosystem component has not been met?
Figure 8. The draft decision process.

References

Figure 9. A visual representation of a regional assessment.
ANNEX 4 Birds report
MSFD Targets and Indicators for Marine Birds

Sub-group Chair: Ian Mitchell (JNCC)
Sub-group members: A. Webb (JNCC), A. Brown (NE), A. Douse (SNH), S. Foster (SNH), N. McCulloch (EA-NI), M. Murphy (CCW); additional input from D. Stroud (JNCC).

Introduction

The UK’s marine environment holds internationally important numbers of birds during the breeding season, during spring and autumn migration and during winter. In total, 109 bird species regularly use the marine areas in the UK, chiefly as a source for food but also as a safe place away from land-predators in which to loaf, roost and moult. Most of the UK’s marine bird species come from two broad taxonomic groups that are commonly referred to as waterbirds and seabirds.

There are 57 species of waterbird that regularly use the UK’s marine environment, comprising of shorebirds (order Charadriiformes, including representatives form the following families: - Haematopodidae (e.g. oystercatcher), Recurvirostridae (e.g. curlew), plovers - Charadriidae and sandpipers - Scolopacidae); herons, egrets and spoonbills (Ciconiiformes: Ardeidae and Threskiornithidae); ducks, geese and swans (Anseriformes); divers (Gaviiformes); grebes (Podicipediformes); and coot (Gruiformes: Rallidae). Shorebirds and some duck species feed on benthic invertebrates in soft inter-tidal sediments and on rocky shores. Geese mostly graze on exposed eelgrass beds (i.e. Zostera spp.). Herons and egrets feed on fish in shallow (i.e. wading depth) inter-tidal and sub-tidal areas, whereas grebes, divers and some duck species can catch fish in deeper water. Other diving ducks feed on invertebrate benthos.

However, the coastal area under the jurisdiction of MSFD includes only non-estuarine shores below MHWS that could include lagoons and saltmarsh that are not associated with transitional waters (i.e. places that are inundated at high tide without flow of freshwater). Therefore, in determining GES under MSFD with respect to marine birds, we need only to focus on those species of shorebird and other waterbirds that predominate outside estuaries.

There are 38 species of seabird that regularly occur in the seas around the UK. They comprise of Procellariiformes: fulmars and shearwaters (Procellariidae) and storm-petrels (Hydrobatidae); Pelecaniformes: gannets (Sulidae ), great cormorant and European shag (Phalacrocoracidae); and Charadriiformes: skuas (Stercorariidae), gulls (Laridae), terns (Sternidae ) and auks (Alcidae). Most seabirds feed on prey living within the water column (i.e. plankton, fish and squid) or pick detritus from the surface. Gulls are the only seabirds that also feed on benthos, by foraging along exposed inter-tidal areas. Therefore most seabirds spend the majority of their lives at sea: some are confined to inshore waters (e.g. terns, gulls, great cormorant and European shag) and others venture much further offshore and beyond the shelf-break, even during the breeding season.

3 According to JNCC’s list of bird species that make significant use of the marine environment around the UK – see www.jncc.gov.uk/page-4560.
Relevant Commission Decision Criteria / Indicators

Table 1 shows the Commission Decision criteria and indicators that are relevant to birds under Descriptor 1: Biological Diversity and Descriptor 4: Food Webs. The table does not include the indicators listed in the Commission Decision criteria 1.4, 1.5 and 1.6 under Descriptor 1, that are relevant only to marine habitats.

Indicators and targets have been proposed for each commission Decision indicator that are considered relevant. No indicators or targets are proposed under the Commission Decision indicator of population genetic structure (1.3.2). But there should be an assessment of genetic structure in harbour seal populations to enable indicators of population abundance (1.2.1) to be equally representative of all discrete population sub-units. There are no genetically distinct sub-populations within any of the species of birds using UK waters that would benefit from an indicator or target, in order to maintain its diversity.

Bird functional groups, listed species and indicator species

The recommended functional groups for birds (Table 2) are identical to those recommended by OSPAR’s MSFD advice manual on biodiversity (OSPAR Commission unpub.).

Within each functional group, the species that should be included in UK assessments of GES were selected according to following groupings as specified in the Directive:

i. Listed species from Community legislation and international agreements (hereafter, referred to as ‘listed species’).

ii. Additional species being considered within some [European] sub-regions for potential use to represent the broader functional group in which they occur (hereafter, referred to as ‘indicator species’).

The selection of marine bird species should be limited to those that occur regularly in the MSFD assessment area. This excludes some waterbird species that predominate in estuaries.

OSPAR Commission (unpub.) recommends ‘listed species’ of birds should be those that are included in Annex 1 of the Birds Directive and on the OSPAR list of threatened and declining species. The Birds Directive actually applies to all wild migratory bird species and Annex 1 lists those species for which nationally important aggregations should be designated as Special Protection Areas, as oppose to internationally important aggregations in all other species. Hence, the Birds Directive is not a necessarily a useful reference for identifying species that require special protection and inclusion in assessments of GES under MSFD. Furthermore, the OSPAR list of threatened and declining species does not appear to be inclusive of all relevant taxa of marine birds. Therefore, we recommend that ‘listed species’ are selected from the species that are awarded the highest level of protection under the Action Plan of AEWA - African Eurasian Waterbird Agreement (i.e. species listed in column A of Table 1, Annex III of the Agreement).

However, AEWA does not include petrels and shearwaters. However, if the AEWA criteria (appended to this Annex 4) were applied to the petrels and shearwaters that regularly occur in UK waters, Balearic shearwater would be added to the list.

The selection of bird ‘indicator species’ was guided by OSPAR Commission (unpub.) that provides the following guidance: The selection of species to be assessed under MSFD in the OSPAR maritime area (MSFD subregion b) should be representative in terms of:
a. their abundance and distribution (i.e. also naturally predominant species as well as species that are predominant as an effect of human activities should be included);

b. their sensitivity towards specific human activities;

c. their suitability for the respective indicators and descriptors of the EU COM decision;

d. the practicability (incl. cost effectiveness) to monitor them;

e. their inclusion in existing monitoring programmes and time-series data;

f. their association with specific habitats.

The full list of proposed indicator species of marine bird in UK waters is embedded here:

The use of seabirds and waterbirds that are monitored at sea as indicator species is very much dependant on the development of new monitoring and its ability to provide accurate data at a suitable spatial scale that can be incorporated into the indicators recommended below.

**Indicators, Baselines & Targets:**

**Approach to setting baselines and targets for marine bird indicators**

For birds, the limited time series of data available (around 30-40 years) do not contain any true reference values when anthropogenic influence on these animals was negligible. It is unlikely that populations of these highly mobile animals have at no part in their lives not been impacted by current or past human influences on the marine environment. Hence there are no reference populations that state targets can be set against Baselines set in the past or current values in a time series are likely to be biased by certain anthropogenic impacts. As measures are implemented (e.g. under MSFD) to reduce these impacts, the baseline values set in the past may no longer be appropriate for assessing GES. For example, some seabird species are arguably more abundant as a result of food provided by wasteful fishing practices such as discarding. As the CFP moves towards eliminating discards, these species may decline in number and range beyond target levels and therefore fail to achieve GES. Baselines for mobile species should therefore be reviewed during every 6 year reporting cycle and amended if necessary to take account of any anthropogenic bias in previously set baselines.

Baselines should also be reviewed regularly to ensure they take into account the inherent variance in the marine environment, and to ensure that the state of indicators at GES with respect to Descriptor 1 is in accordance with ‘prevailing geographic, physiographic and climatic conditions’. As monitoring programmes develop and evidence increases, our understanding of the likely future impacts of a changing environment will enable us to set ‘smarter baselines’ that can account for such changes.

**Species Distribution (Criterion 1.1)**

With respect to Species distribution (Criterion 1.1), separate indicators (and indicator-targets) of distributional range (1.1.1) and distributional pattern (1.1.2) are proposed for each of five groups of birds:
i. breeding seabirds – refers to all seabird species at breeding colonies;

ii. coastal-breeding waterbirds – refers to all species of waterbird breeding close to the shoreline and dependant on intertidal and inshore areas for feeding;

iii. non-breeding waterbirds – refers to all waterbird species in inshore waters outside the breeding season;

iv. non-breeding shorebirds – refers to all species of in non-estuarine intertidal areas outside the breeding season;

v. seabirds at sea – refers to all seabird species in inshore and offshore waters throughout the year.

The separation is due to differences in how the range and distributional pattern of each group will be estimated. No indicator is proposed for distributional range (1.1.1) for seabirds at sea, since the extremities of the range of such wide-ranging species are unlikely provide an indicator of GES. More pertinent to GES of seabirds at sea will be an indicator of distributional pattern (1.1.2) that will detect changes in the distribution of high and low density areas that may be linked to the distribution and intensity of pressures.

None of the indicators for range and distributional pattern in birds are currently operational because there is no other requirement for such indicators. Sufficient data do exist for breeding seabirds, coastal-breeding shorebirds and non-breeding shorebirds to enable indicators to be defined by 2012 and be operational by 2014. The construction of indicators for non-breeding waterbirds and seabirds at sea, is subject to the progress of the UK Seabird & Cetacean Monitoring Project, which is currently under development. There is currently no systematic monitoring of inshore aggregations of waterbirds in UK waters. Seabird monitoring in the UK is limited mainly to providing information at colonies, and it is likely that additional monitoring will be required in offshore and inshore waters as part of the EU Birds Directive. The monitoring of seabirds at colonies provides the most cost-effective and precise information on their numbers and distribution compared to other monitoring methods. However, there are limitations in this approach to understanding how human pressures are affecting any measured changes; the monitoring does not occur at the location of the impacts in offshore waters and fluctuations at the colonies will be affected by changes in all of the areas used for feeding and maintenance by seabirds, not just the areas where the human pressures are occurring. Furthermore, monitoring at colonies provides no information on how the distributional range and population size of breeding species changes in UK waters outwith the breeding season, and provides no information at all for species that do not breed in the UK but visit our waters in important numbers during the non-breeding periods. Monitoring of the UK’s waters for these top predators is likely to utilise information from a wide variety of sources in order to make surveys cost-effective, such as from volunteers, marine industries supplemented by targeted information commissioned to fill remaining gaps in data provision from. Data collection surveys can be combined for seabirds and for cetaceans in most cases at no additional cost. If sufficient monitoring of inshore waterbirds and seabirds at sea is instigated as part of the project, then indicators and targets for these birds could be operational by 2018; this also applied to indicators of population size.

4 From the Seabird Monitoring Programme (SMP), Wetland Bird Survey (WeBS), Non-estuarine Wader Survey (NEWS), successive breeding and wintering Bird Atlases and successive breeding seabird censuses (e.g. Seabird 2000) – all covering the British Isles.
Population Size (Criterion 1.2)

The indicators proposed for bird population size (Criterion 1.2) are more generic than for distribution: i) Species-specific trends in relative breeding abundance and ii) Species-specific trends in relative non-breeding abundance. The metrics for both are annual estimates of the numbers of birds (or pairs) expressed as a proportion of a baseline value that is specific to each species. The two indicators provide a functional distinction in the case of waterbirds, between state of populations during breeding and at other times; in the case of seabirds between the state of breeding seabirds at colonies and the state of populations at sea.

The recommended approach to indicators (and targets) of bird population size follows that developed by ICES (2008) for the OSPAR EcoQO on seabird population trends. Existing UK and devolved administration indicators for populations of breeding seabirds and non-breeding waterbirds use a single trend of the annual geometric mean abundance of multiple species with each group (e.g. http://jncc.defra.gov.uk/page-4229). ICES (2008) considered using geometric mean trend indicator for the EcoQO on seabird population trends but discounted it on the basis that it was difficult to interpret and difficult to set targets against and use to inform measures.

Data exist on abundance of breeding seabirds, coastal breeding waterbirds and non-breeding shorebirds and will be collected by ongoing schemes (though some expansion may be necessary – see below). A breeding seabird indicator has already been produced for the Celtic Seas (OSPAR Region III – see Figure 1 taken from see ICES 2010) and one is currently being developed with neighbouring states for the Greater North Sea (OSPAR Region II). Population size indicators for the aforementioned groups of bird should be operational by 2012.

Setting baselines and targets for population size (1.2) should follow the approach used in the OSPAR draft EcoQO on seabird population trends (see ICES 2008, 2010). Baselines for each indicator of population size (criterion 1.2) should be set in the past at a time when anthropogenic influence on a particular species was thought to be minimal. Indicator targets should be set as a deviation from the baseline and the target can be varied according to the ability of species to recover from declines: ICES (2008) recommended that annual abundance of species that lay more than one egg per year should be more than 70% of the baseline, but more than 80% of the baseline for species that lay just one egg. The different targets are intended to take account of the lower reproductive output of the single egg layers and hence, a lower rate of recovery following declines. Targets for positive deviations should be set, where population increase may have an impact on GES of the wider marine bird community (e.g. species that depredate other birds and benefit from anthropogenic food sources). ICES (2008) recommended an upper limit of +30% of the baseline for all species. However, the HBDSEG birds subgroup recognised that increases in some species may have negligible effects on the rest of the seabird community and if such species were to exceed an upper target value, this may not necessarily mean that GES is not achieved. Therefore, upper limits to indicator targets should be applied only to those species (i.e. skuas and large gulls) whose populations may be artificially elevated by anthropogenic impacts (e.g. food provided by discarding) and would also, in turn, have a detrimental impact on other bird populations.

The criterion targets for species distribution and population size should be set at a proportion of species-specific indicators that are within target values: ‘Changes in abundance should be within target levels for >75% - >90% of species of marine bird that are monitored’. The lower limit of 75% is taken from the OSPAR draft EcoQO on seabird population trends (ICES 2008): if less than 75% of species indicators had not achieved their target, the seabird community would possibly be in poor health and suitable action would be instigated (i.e. research or remedial measures). Given that ICES (2008) considered 75% to be the limit below which
remedial action should be instigated, the option of a higher target, up to 90%, is more likely to equate to GES. Figure 2 shows an example of this criterion for breeding seabirds in the Celtic Seas OSPAR Region III – from ICES (2010).

**Population Condition (Criterion 1.3)**

For Population Condition (Criterion 1.3) we have proposed indicators of demographic characteristics (1.3.1) for species that breed in the UK, for which metrics such as breeding success and adult survival are relatively straightforward to monitor compared to those species that occur in UK waters outside the breeding season. Appropriate indicator species are predominantly seabirds but some waterbirds such as common eider may be appropriate.

Breeding success is one of the recommended metrics as this is already monitored annually in some seabird species at colonies throughout the UK, with a time series going back to the mid 1980s. We propose two indicators of breeding success:

i. Annual breeding success of kittiwakes

ii. Breeding failure of seabird species sensitive to food availability

Some further development work is required for both indicators but both should be operational by 2012.

The baseline-setting method used for indicators of Population condition (1.3) and productivity (4.1) vary depending on indicator used. For the indicator on breeding failure of seabird species sensitive to food availability, there is no baseline, whereas the indicator of annual breeding success of kittiwakes, has a baseline that takes into account climatic variation. Kittiwake breeding success at individual colonies is significantly negatively correlated with local mean sea-surface temperature (SST) two winters previously (Frederiksen et al. 2004, 2007). The relationship is thought to be related to larval sandeel survival and the subsequent availability of 1 year-class sandeels for kittiwakes to rear their chicks on. Therefore at each colony, a different baseline value is set each year according to the local SST two winters previously.

In an area of E Scotland and NE England where sandeel fishing occurred during 1991-98 and has been banned since 2000, there was a significant additive effect of the presence of fishing, as shown the red line in Figure 3.

The proposed indicator uses the regression of SST and breeding success as the baseline and the 95% CI as the target ‘band’ (see Figure 3). It is proposed that any significant negative deviation will indicate a detrimental anthropogenic impact other than any climate change impacts. Breeding success should be within the target band in at least five years in each six year reporting cycle, in order to allow for natural stochastic events that may depress breeding success (e.g. heavy rainfall). The criterion target should be set at the number of colonies achieving the target rate (e.g. 50, 75 & 90%, but not necessarily these values).

JNCC plan further work to look at whether baselines and targets can be constructed for other colonies elsewhere in the UK and whether other species that are sensitive to food supply could be included.

We identified a need to develop an indicator for pressures on land that would significantly affect the productivity of seabirds as marine organisms. Depredation by non-native mammals (e.g. rats, mink) on islands can reduce breeding success, breeding numbers and extirpate colonies. The pressure from non-native mammals can be easily removed through eradication
and quarantine. If this pressure is removed it will mitigate the impacts of other pressures that are not so easy to manage e.g. climate change and fishing pressures.

The other demographic measured in seabirds is adult survival. However, annual survival rates are currently monitored in a handful of colonies per species. A project by JNCC and the BTO is underway to investigate the feasibility of expanding existing monitoring. An indicator on adult survival would be dependent on future monitoring, but may be operational by 2018.

There may be problems linking changes in survival with pressures operating in UK and European waters, since many species spend the non-breeding period in areas beyond the NE Atlantic. But we suggested that an indicator of birds killed by commercial fishing (‘bycatch’) and aquaculture should be developed to monitor a potentially significant pressure on marine bird survival. Monitoring of seabird bycatch is conducted in UK waters but it is incidental and not part of systematic survey and excludes some gear-types and aquaculture. As a consequence, the extent of this pressure in UK waters is unclear, but could potentially kill large numbers of birds. Further monitoring is required in order to determine how important this pressure is on marine birds and to develop an indicator.

Indicators of population condition (1.3) and population size (1.2) described above, should be used as indicators of Productivity (production per unit biomass) of key species or trophic groups (Criterion 4.1) and of Abundance/distribution of key trophic groups/species (Criterion 4.3)

**Ecosystem structure (Criterion 1.7)**

We envisage that some of the indicators proposed for 1.1, 1.2 and 1.3 will contribute to this criterion but did not discuss their inclusion in anymore detail.

**Productivity (production per unit biomass) of key species or trophic groups (Criterion 4.1)**

See productivity indicators and targets under Criterion 1.3.

**Abundance/distribution of key trophic groups/species (Criterion 4.3)**

See population size indicators and targets under Criterion 1.2.

**References**


**Appendix - AEWA Classification used in the context of MSFD to define species listed under the Agreement**

**STATUS OF THE POPULATIONS OF MIGRATORY WATERBIRDS**

**KEY TO CLASSIFICATION**

The following key to Table 1 is a basis for implementation of the Action Plan:

**Column A**

**Category 1:**

a) Species, which are included in Appendix I to the Convention on the Conservation of Migratory species of Wild Animals;

(b) Species, which are listed as threatened on the IUCN Red list of Threatened Species, as reported in the most recent summary by BirdLife International; or

(c) Populations, which number less than around 10,000 individuals.

**Category 2:** Populations numbering between around 10,000 and around 25,000 individuals.

**Category 3:** Populations numbering between around 25,000 and around 100,000 individuals and considered to be at risk as a result of:

(a) Concentration onto a small number of sites at any stage of their annual cycle;

(b) Dependence on a habitat type, which is under severe threat;

(c) Showing significant long-term decline; or

(d) Showing extreme fluctuations in population size or trend.

For species listed in categories 2 and 3 above, see paragraph 2.1.1 of the Action Plan contained in Annex 3 to the Agreement.

Source:

Table 1 Relevance of Commission Decision criteria and indicators to marine birds.

**RIT** - relevant and indicator and target proposed, **RI** - relevant and indicator only proposed, **R** – relevant but no target or indicator proposed, **X** - not relevant.

<table>
<thead>
<tr>
<th>Criterion</th>
<th>Indicator</th>
<th>Relevance to Marine Birds</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.1 Species distribution</td>
<td>Distributional range (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)</td>
<td>RIT</td>
</tr>
<tr>
<td>1.2 Population size</td>
<td>Population abundance and/or biomass, as appropriate (1.2.1)</td>
<td>RI</td>
</tr>
<tr>
<td>1.3 Population condition</td>
<td>Population demographic characteristics (e.g. body size or age class structure, sex ratio, fecundity rates, survival/mortality rates) (1.3.1), Population genetic structure, where appropriate (1.3.2),</td>
<td>R</td>
</tr>
<tr>
<td>1.7 Ecosystem structure</td>
<td>Composition and relative proportions of ecosystem components (habitats and species) (1.7.1),</td>
<td>R</td>
</tr>
<tr>
<td>4.1 Productivity (production per unit biomass) of key species or groups</td>
<td>Performance of key predator species using their production per unit biomass (productivity) (4.1.1),</td>
<td>R</td>
</tr>
<tr>
<td>4.2 Proportion of selected species at the top of food webs</td>
<td>Large fish (by weight) (4.2.1),</td>
<td>X</td>
</tr>
<tr>
<td>4.3 Abundance/distribution of key groups/species</td>
<td>Abundance trends of functionally important selected groups/species (4.3.1)</td>
<td>RIT</td>
</tr>
</tbody>
</table>
### Table 2  Recommended functional groups and listed species for marine birds in UK waters.

<table>
<thead>
<tr>
<th>Functional Groups</th>
<th>example taxa or group</th>
<th>Listed species</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inshore pelagic feeding birds</td>
<td>divers, grebes, saw-bills, great cormorant, European shag, black guillemot</td>
<td>Great Northern diver&lt;sup&gt;1&lt;/sup&gt;</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Slavonian grebe&lt;sup&gt;1&lt;/sup&gt;</td>
</tr>
<tr>
<td>Inshore surface feeding birds</td>
<td>gulls, terns</td>
<td>Little Tern&lt;sup&gt;1&lt;/sup&gt;</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Roseate tern&lt;sup&gt;1,2&lt;/sup&gt;</td>
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<tr>
<td>Inshore benthic feeding birds</td>
<td>seaduck</td>
<td>Smew&lt;sup&gt;1&lt;/sup&gt;</td>
</tr>
<tr>
<td>Intertidal benthic feeding birds</td>
<td>waders</td>
<td>Purple sandpiper&lt;sup&gt;1&lt;/sup&gt;</td>
</tr>
<tr>
<td>Offshore pelagic feeding birds</td>
<td>auks, northern gannet, shearwaters</td>
<td>Balearic shearwater&lt;sup&gt;2&lt;/sup&gt;</td>
</tr>
<tr>
<td>Offshore surface feeding birds</td>
<td>black-legged kittiwake, northern fulmar, storm-petrels</td>
<td>Black-legged kittiwake&lt;sup&gt;2&lt;/sup&gt;</td>
</tr>
</tbody>
</table>

<sup>1</sup>AWEA Annex1 Table 1 column A

<sup>2</sup>OSPAR List of threatened and declining species.
Figure 1 Trend in relative abundance of European shag in OSPAR Region III Celtic Seas. Indicator target was set at +/- 30% of the baseline. (From ICES 2010)

Figure 2 The EcoQO on seabird population trends applied to data on breeding seabird trends (as in Fig. 1) from OSPAR Region III Celtic Seas. Target was set at >75% of species should be at target levels for abundance (From ICES 2010)
Figure 3 Stylised version of a relationship between kittiwake breeding success and SST two winters previously (from Frederiksen et al. 2004) to demonstrate how targets may be set re Criterion 1.3.
ANNEX 5 Marine mammals report

MSFD Marine Mammal and Reptile subgroup: Criterion Targets audit trail

Subgroup Co-Chairs: Eunice Pinn (JNCC) and Mark Tasker (JNCC)

Subgroup Members and contributors: Phil Hammond (SMRU), Callan Duck (SMRU), Alisa Hall (SMRU), Clare Ludgate (NE), Fiona Manson (SNH), Karen Hall (SNH), Jim Reid (JNCC), Victoria Copley (NE), Mandy McMath (CCW) and Gary Burrows (NIEA).

Background to the species

Cetaceans: Whales, dolphins and porpoises are collectively known as cetaceans. Twenty-eight species have been recorded in UK waters. Of these, 11 are known to occur regularly, while the remaining 17 species are considered to be vagrants or rare visitors. This represents a high level of cetacean diversity within the UK’s comparatively small section of the North Atlantic, and is due to the considerable diversity in topography, habitats and food resources available in these waters.

Cetaceans are very mobile and can range widely, with some undertaking large-scale movements including regular seasonal migrations while others display more localised movements. For most species, animals found in UK waters are therefore part of a much larger biological population or populations whose range extends beyond UK waters. Equally, the number of individuals present at any one time may be only a small proportion of those that make use of UK waters at some point during their lives.

Seals: Two species of seal live and breed in UK waters, grey seals (Halichoerus grypus) and harbour (also called common) seals (Phoca vitulina). Although both species can be found around the UK coast at any time of the year, they are not evenly distributed. Both are considerably more abundant in Scotland than in England, Wales or Northern Ireland; with harbour seals being rare on the south and west coasts of England and in Wales.

A number of Arctic seals are seen very occasionally around the UK coast, particularly in Scotland, including ringed (Phoca hispida), harp (Phoca groenlandica), bearded (Erignathus barbatus) and hooded seals (Cystophora cristata), and very rarely walrus (Odobenus rosmarus). These species are not resident in the UK and are not further considered.

Turtles: Encountering a marine turtle in UK waters is a rare event; this is partly due to their relatively low numbers and partly to the difficulty of spotting them in the open sea when conditions are anything but perfectly calm. Of the 4 species reported from UK waters, the leatherback turtle (Dermochelys coriacea) is the only one to be considered a true member of the British fauna; with approximately 30 records per year. UK waters represent only a small peripheral part of the foraging habitat of this wide-ranging species. All other turtle species (loggerhead Caretta caretta, Kemp’s ridley Lepidochelys kempii and green Chelonia mydas) tend to reach UK

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waters only when displaced from their normal range by adverse currents, i.e. UK waters are not considered part of their functional range.

Because these species are either not resident in the UK or are seen only very rarely, they are not considered to be appropriate indicator species.

**Development of GES Criterion Targets**

All marine mammal species in the UK are listed under a variety of Community Legislation and international agreements: the Habitats Directive, Convention on Migratory Species and ASCOBANS - Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas, and OSPAR list of threatened and declining species. As such, all marine mammal species in UK waters are considered to be ‘listed species’.

The marine mammal and reptile subgroup have proposed a total of 41 indicators and targets, mostly species specific, which link to 3 Criterion Targets for Descriptor 1 biodiversity, 2 Criterion Targets for Descriptor 4 Food webs and 3 additional Criterion Targets linked to pressure impacts. Indicators and targets were only proposed for the most commonly seen species, i.e. harbour and grey seals, harbour porpoise, short-beaked common dolphin, bottlenose dolphin, white-beaked dolphin, long-finned pilot whale and minke whale.

These indicators and targets all link to assessments that are integral to the UK’s obligations through the Habitats Directive, ASCOBANS, OSPAR and EU Regulation 812/2004 on cetacean bycatch.

For the consultation, Defra have proposed that there will be three levels of Criterion target:

- **Level 1**: GES targets and indicators which are well developed and supported by good evidence and which the UK could put forward to the Commission in 2012.
- **Level 2**: GES targets and indicators which need some further development work between now and 2012 in order to finalise the details and Defra propose using the consultation to seek people’s views on the suggested criterion target.
- **Level 3**: GES targets and indicators for which it has not been possible to develop proposals at this time. With further research and developmental work, it may be possible to put these forward in 2018.

**Descriptor 1: Biodiversity**

**Criterion Target 1.1 Species Distribution**: In 75-90% of indicators monitored, there should be no statistically significant contraction in the distribution of marine mammals.

Four of the indicators and targets, related to seals, proposed under this criterion target are considered to represent level 1, i.e. they are well developed and there is sufficient evidence for them to be submitted in 2012. There are an additional 6 relating to cetaceans that require further research and developmental work. Research is currently being undertaken that will, by March 2012, provide a good indication of our ability to detect trends in cetacean distribution and abundance for the more common species, including the power to detect those trends through data collected under the Joint Cetacean Protocol. Once this information is available, it is hoped that these indicators and targets will contribute to the Criterion Target. This will also
rely on the development of a cetacean monitoring programme as required by the Habitats Directive. Such a programme has yet to be fully developed, costed and implemented.

For seals, this criterion target links to work undertaken for the two OSPAR EcoQOs. The annual assessment of UK seal populations is based primarily on information from surveys conducted by the Sea Mammal Research Unit of both grey and harbour seals in Scotland and of common seals in eastern England (Figure 1 and 2) but includes information collected independently by other organisations, including: the National Trust (Farne Islands and Blakeney Point), Scottish Natural Heritage (Shetland, Lismore, South Ronaldsay and Rum), Lincolnshire Wildlife Trust (Donna Nook), Natural England (Horsey) and the Countryside Council for Wales (Wales). In Northern Ireland, the Northern Ireland Environment Agency (formerly Environment and Heritage Service) and the National Trust monitor common and grey seals Strangford Lough. A small number of other organisations also collect information on grey and common seals (e.g. INCA study seals in the estuary of the River Tees; the Cornwall Seal Group studies seals in Cornwall and the Isles of Scilly). Information from these organisations may be included into the formal advice provided through the Special Committee on Seals where appropriate. The UK seal monitoring programme is largely funded by NERC and costs approximately £1M per annum. However, this is supplemented by funding, particularly for harbour seals from SNH and Natural England and from Scottish Government. NERC have recently indicated that their funding will be reduced by up to 20% over the next 3-4 years which is likely to mean a reduction in the number of sites surveyed and/or frequency of surveys, particularly for grey seals.
Figure 1. The main grey seal breeding colonies in Great Britain and Northern Ireland (From SCOS, 2008).
Figure 2. The distribution of common seals in Great Britain and Northern Ireland in August, by 10km squares, from surveys carried out between 2000 and 2006 (From SCOS, 2008).
Criterion Target 1.2 Population Size: In 75-90% of indicators monitored, there should be no statistically significant decrease in abundance of marine mammals.

Two of the indicators and targets contributing to this criterion target are considered to be sufficiently well developed for submission in 2012, these relate to seals and are derived from two OSPAR EcoQOs.

For grey seals:
"Taking into account natural population dynamics and trends, there should be no decline in pup production of grey seals of ≥10% as represented in a five-year running mean or point estimates (separated by up to five years) within any of nine sub-units of the North Sea. These sub-units are: Orkney; Fast Castle/Isle of May; the Farne Islands; Donna Nook; the French North Sea and Channel coasts; the Netherlands coast; the Schleswig-Holstein Wadden Sea; Heligoland; Kjørholmane (Rogaland)."

And for harbour seals:
"Taking into account natural population dynamics and trends, there should be no decline in harbour seal population size (as measured by numbers hauled out) of ≥10% as represented in a five-year running mean or point estimates (separated by up to five years) within any of eleven sub-units of the North Sea. These sub-units are: Shetland; Orkney; North and East Scotland; South-East Scotland; the Greater Wash/Scroby Sands; the Netherlands Delta area; the Wadden Sea; Heligoland; Limfjord; the Kattegat, the Skagerrak and the Oslofjord; the west coast of Norway south of 62°N."

In the UK, up to January 2009, grey pup production remains within the limits of the EcoQO (Figure 3). Pup production appears to be beginning to stabilise in Orkney; is increasing at the Isle of May/Fast Castle (due entirely to increases at Fast Castle); is stable at the Farne Islands and is increasing at Donna Nook. Two colonies recently established in Norfolk (at Blakeney Point and at Horsey) should be included with Donna Nook in this EcoQO assessment.

In contrast, recent surveys have shown that harbour seal populations have declined well in excess of the limits set out in the EcoQOs in Orkney, Shetland, north-east Scotland, south-east Scotland and the Greater Wash (Figure 4). The decline in the east of England was due to the outbreak of PDV in 2002. Reasons for declines in the other areas have not yet been determined.

There are an additional 7 indicators and targets relating to cetaceans that require further research and developmental work to enable an assessment of the Criterion Target. See comments in criterion target 1.1 above regarding development of the first 6 through work on the Joint Cetacean Protocol. The 7th relates to the inshore bottlenose dolphin populations. For two of these populations (Scottish east coast and the Cardigan Bay area), there is the potential that such an indicator could be developed in time for 2012, although 2018 is considered more realistic. There is sufficiently reliable data to indicate that the bottlenose dolphins on the Scottish east coast is considered to be stable and those in the Cardigan Bay area also remain stable, with limited evidence of a slight increased (Figures 5 and 6). These abundance estimates were collected during research projects were funded in part by SNH and CCW, respectively. SNH have an MoU in place with Aberdeen University to continue to carry out this survey work, which runs until 2013. Funding for the work in Wales has recently
ceased and now relies solely on funds that can be raised by Sea Watch Foundation (who were originally contracted by CCW to undertake the work).

Figure 3. Grey seal pup production at annually monitored colonies in Scotland and England between 1960 and 2007 (From SCOS, 2008). These colonies account for approximately 85% of grey seal pups born in the UK.
Counts of harbour seals around Scotland

Figure 4. Counts of common seals in Regions of Scotland between 1990 and 2007 (From SCOS, 2008).

Figure 5: Trends in annual estimates of the number of dolphins using the Moray Firth SAC, based upon surveys conducted during the core-study inner Moray Firth study area (From Thompson et al., 2006).
Figure 6: Bottlenose dolphin population estimates from photo-ID work in the Cardigan Bay area (From Peasante et al, 2008). Error bars represent standard error.

Criterion Target 1.3 Population Condition: There should be no statistically significant decline in seal pup production and bottlenose dolphin calf production; and there should be no adverse health effects from contaminants and biotoxins; and mortality of marine mammals due to fishing bycatch should be sufficiently low to not inhibit population size targets being met.

A total of 11 indicators and targets currently contribute to this Criterion Target. Of these 2 are sufficiently well developed, 5 require some additional development but will be ready for 2012 and a further 4 are require further research with the possibility of submission in 2018. These 11 indicators and targets cover a variety of biological aspects:

- pup and calf production for the two seal species and the inshore bottlenose dolphin populations, respectively (see section on Criterion 4.1).
- bycatch of harbour porpoises, short beaked common dolphins, grey seals and harbour seals (see section on MSFD pressure indicators)
- PCB and other organohalogenated contaminant levels in harbour porpoises and harbour seals linked immunosupression and other adverse health effects (see section on MSFD pressure indicators)
- the relative occupancy of haulout sites used by both grey and harbour seals. Additional analysis underway to determine if there is a relationship between relative haulout numbers and local trends in population abundance between the two species. Existing data will need to be analysed to provide an indication of feasibility of such an assessment.
- identification of harbour seal subpopulations. The assessment of genetic structure of this species will enable estimates of the abundance of harbour seal in discrete population sub-units to be made, which is considered extremely important due the continuing population declines observed.
- presence of algal biotoxins in seals (see section on MSFD pressure indicators).

Many of these proposed indicators are reliant on short-term research funds awarded to academic institutions or NGOs.
Descriptor 4: Food webs

Indicators proposed for marine mammals under D4 criteria 4.1 and 4.3 are identical to those indicators proposed above under Criteria 1.2 and 1.1 respectively.

Criterion Target 4.1 Productivity (production per unit biomass) of key species or trophic groups: In 75-90% of indicators monitored, there should be no statistically significant decrease in abundance of marine mammals.

This covers the pup and calf production for the two seal species and the inshore bottlenose dolphin populations, respectively. For grey seals there is sufficient evidence to submit in 2012 (e.g. see Figure 4). For harbour seals and the inshore bottlenose dolphin populations some additional work is required, but the indicators are expected to be submitted in 2012.

Harbour seal pup production is difficult to assess. It has, however, been monitored annually in the Moray Firth since 1988, although more accurate aerial techniques were introduced in 2006. Assessments are also undertaken in The Wash, where 90% of the species outside Scotland resides. The Wash was first surveyed in 2003 and is reassessed every 5 years. Further sites (possibly SACs) should be added to these assessments, but will require additional funding.

Photo-identification studies in the Cardigan Bay area indicate that the number of calves born per year are 13 (2005), 20 (2006), and 20 (2007). From those estimates, crude birth rates were calculated as 0.098 (2005), 0.112 (2006), and 0.101 (2007), and a mean value for the three years of 0.104. These compare favourably with crude birth rate estimates calculated for other bottlenose dolphin populations, which range from 0.012 to 0.156, but in most cases are between 0.055 and 0.10. Until recently this work undertaken by the Sea Watch Foundation, funded by CCW. These funds have now ceased and continuation of the research relies on monies obtained by Sea Watch. Similar photo-ID work has been undertaken in the Moray Firth through CRRU and Aberdeen University.

Criterion Target 4.2 Abundance/distribution of key trophic groups/species: There should be no statistically significant decline in seal pup production and bottlenose dolphin calf production.

Two indicators contributing to this Criterion target are sufficiently developed (grey and harbour seal abundance trends) with a third expected by 2012 (inshore bottlenose dolphin abundance trends). The development of an additional 6 indicators linked to cetacean abundance and another for the relative occupancy of seal sites is expected by 2018.

At this time, trends in distribution of these marine mammal species have been left out of this Criterion target. However, should the ongoing JCP research indicate that value would be gained with their inclusion then the number of indicators contributing to this Criterion target will be increased.
MSFD Pressure targets associated with marine mammals

Three GES Criterion Targets have also been suggested for pressures known to have potentially significant influence on marine mammals populations. These cover bycatch, PCB contamination and algal biotoxins.

**MSFD Pressure Impact selective extraction of species, including incidental non-target catches (e.g. by commercial and recreational fishing: Annual bycatch rate is reduced to less than 1.7% of best population estimate (for harbour porpoise and common dolphin, but 2% for seals)**

At the fifth North Sea Conference in 2002, Ministers agreed that an ecological quality element relating to harbour porpoise bycatch in the North Sea would be given the objective: ‘annual bycatch levels should be reduced to levels below 1.7% of the best population estimate’. In 2006, OSPAR adopted the agreement on the application of the ecological quality objective (EcoQO) system in the North Sea, which required in 2008, a first assessment of the application of the EcoQO system and in 2009 an improved elevation of the results of the EcoQO system as a contribution to the Quality Status Report (QSR) for 2010 (OSPAR, 2010).

In 2008, the ICES (International Council for the Exploration of the Sea) Working Group on Marine Mammal Ecology tried to evaluate progress to date with this EcoQO on a North Sea wide basis (ICES, 2008b). It was quickly apparent that many of the fisheries suspected to have the highest bycatch levels are conducted without bycatch observer programmes as these are not a requirement of Council Regulation 812/2004 (see below). As a consequence, it was not possible to evaluate whether the EcoQO has been met.

Council Regulation 812/2004 requires the reporting of certain cetacean bycatch from all EU Member States, although these are not fully comprehensive of the North Sea. In addition, evaluation of the scale of incidental killing and capture of cetaceans (i.e., bycatch) is also required under the EU Habitats Directive, but precise standards have not been set and there has been little evaluation or enforcement of this Directive requirement to date.

For the UK, data on marine mammal bycatch has been formally collected through the UK bycatch monitoring project since 2005. Prior to this data was collected through a variety of research projects. The two main species affected by fishing in UK waters are the harbour porpoise and the short-beaked common dolphin. Harbour porpoises are bycaught mainly in static nets whilst common dolphins tend to be caught in pelagic trawls but are also bycaught in static nets.

UK fleet bycatch estimates of porpoises in the North Sea, though reliant on rather old observations, are estimated to be in the low hundreds at present. These can be compared with notional bycatch limits of around 3500-4500. Other major gillnet fisheries exist in Denmark and Norway, while Sweden, Belgium, the Netherland and France also prosecute gillnet fisheries in The North Sea. Bycatch rates in these other nations’ fisheries are not known at present, but a recent analysis by ICES suggested that total effort in commercial fisheries is not currently high enough to take as many as 3500 porpoises per year in the North Sea, though concerns have been raised about large scale recreational gillnet fisheries that exist along the continental shore of the North Sea. Increasing fuel prices and a potential cod recovery could also increase gillnet effort in the future.
In the South and Southwest, porpoise bycatches are likely in the mid to high hundreds per year in UK set net fisheries, and while sustainable bycatch limits are likely to be over 2000 animals per year, there are as yet very incomplete estimates of bycatch of porpoises in this area by other nations. Similarly, for common dolphins bycatch currently amount to some few hundreds of animals per year taken by the UK fleet, including now just a few or a few tens of animals at most in UK pelagic pair trawl fisheries. The UK total is therefore much lower than the estimated sustainable take limit of around 2800 animals, but bycatch by other European member states are likely to run into the thousands, so it remains a moot point as to whether or not current aggregate bycatch rates for common dolphins are sustainable.

Seals have also been recorded in static nets and pelagic trawls, with bycatch rates highest in the North Sea with, the numbers recorded being relatively low (singles to tens per year).

The UK bycatch monitoring project has recently been awarded for a further 3 years, with the current funding due to cease in 2014. This project covers all cetacean species, seals and seabirds.

**MSFD Pressure Impact.** PCB and other organohalogenated contamination in harbour porpoises and harbour seals are below estimated threshold levels for adverse health effects.

A single indicator contributes to this, which requires further development prior to submission in 2018. Work undertaken through UK cetacean stranding scheme (CSIP) has demonstrated that porpoises dying as a result of infectious disease had significantly higher levels of PCBs than healthy porpoises that die as a result of traumatic deaths (e.g. bycatch or bottlenose dolphin kills) (Jepson et al, 2005). PCB levels equivalent to 13mg/kg lipid has been identified as the critical level at which the contaminant begins to affect cetacean health. Such an assessment is proposed as part of the OSPAR monitoring requirements for harbour porpoises as a ‘threatened and declining species’.

During 2010, Defra funded the analysis of retrospective samples from 100 harbour porpoises (2004-2008) for chlorinated biphenyls (PCBs), organochlorine pesticides (OCs) and brominated diphenyl ethers (flame retardants, PBDEs) with results expected to be available later in 2011, progressing work towards a 20 year time series of marine contaminant analysis in UK stranded harbour porpoises.

In 2010, analyses of long-term temporal trends in blubber concentrations of PCBs (n=440; 1991-2005) (Law et al. 2010a) and PBDEs (n=415; 1992-2008) (Law et al. 2010b) in UK-stranded harbour porpoises were published. Summed PCB concentrations in UK harbour porpoises declined slowly from 1991-1997 and then levelled off in 2005 as a result of a ban on the use of PCBs which began more than two decades ago (Law et al 2010a). This decline is much slower than that observed for organochlorine pesticides (such as DDTs and dieldrin). There are also regional differences in PCBs and OC pesticide levels within UK waters (lower levels in Scotland), possibly reflecting differences in diffuse inputs and transfer between regions, e.g via the atmosphere. The reason for the slow PCB decline is not known but likely to involve continuing diffuse inputs from e.g. PCB-containing materials in storage, construction and in landfills, and to the substantial reservoir of PCBs already in the marine environment.
Among harbour seals comparable datasets exist for blubber levels of PCBs and PBDEs for 1989 and 2003 (Hall et al. 1999, Hall et al. 1992, Hall & Thomas 2007). An archive of blubber samples from the more recent years is also available for analysis. For some regions (e.g. the Wash and the SW coast of Scotland) levels, particularly in males, still exceed the estimated threshold level for adverse health effects, particularly immunosuppressive effects (approximately 20ppm PCBs lipid weight) which have been well demonstrated in this species (Ross et al. 1996).

Funding for this work has so far been ad-hoc. If taken forward, serious consideration will need to be given to finding a more permanent solution to the funding situation. Funding for the UK cetacean stranding scheme itself has recently been put in place for the next three years (approximately £350K per annum) running until 2014. However, this does not include further analysis of toxins either for cetaceans or seals.

**MSFD Pressure Impact Nutrient and organic matter enrichment: Exposure of seals to biotoxins (target to be developed).**

Assessment of toxin levels in seal faeces will provide information on the exposure of seals to the toxins produced by harmful algal blooms which appear to be increasing in many areas throughout the world, including the UK, due to changes in the environment and increases in nutrient input to the marine environment. A single indicator contributes to this, which requires further development prior to submission in 2018. Samples could be obtained through current monitoring of seal diet by the subsampling of faeces collected from seal haulout sites. The adhoc collection of seal scats at various sites around the UK may continue beyond the end of the current diet study and would provide an opportunity to continue the exposure monitoring but some additional funds would be required for the toxin analyses.

**References**


ANNEX 6 Fish report

Developing Indicators and Targets for Descriptors 1 and 4 in Respect of Fish

Data availability and assessment scales

Groundfish surveys have been carried out in support of fisheries management for decades and the species abundance at length data they provide are ideal for deriving the indicators stipulated for Descriptors 1 and 4 in the EC 2010 Decision document. Recent analyses for the UK’s “Charting Progress 2” and OSPAR’s “Quality Status Report 2010” demonstrate that data from such surveys are readily available throughout the majority of UK waters and most waters covering the European continental shelf. The North Sea case study that accompanies this report confirms the wealth of data available to populate these indicators for fish.

Despite this apparent abundance of suitable data, the situation is not perfect. Issues arise regarding the geographic scales at which assessments may be required. The MSFD considers four distinct regions: the Baltic Sea; the North-east Atlantic Ocean; the Mediterranean Sea; and the Black Sea. Two of these regions are further split into four sub-regions. The Greater North Sea (including the Kattegat, and the English Channel) and the Celtic Seas, for example are two sub-regions of the North-east Atlantic Ocean. No single groundfish survey covers an entire MSFD region, and few if any cover an entire sub-region. Assessments at both the sub-regional and regional scale will therefore require aggregation of information collected over smaller spatial sub-units. Protocols detailing how these disparate smaller spatial scale monitoring programmes should be integrated will need to be established so as to ensure that resulting regional-scale assessments are consistent and objective.

Most groundfish surveys are designed to assess fish populations at relatively large spatial scales, for example the North Sea or the Celtic Sea. The North Sea ICES first quarter (Q1) International Bottom Trawl Survey (IBTS) typically samples approximately 170 ICES statistical rectangles and generally achieves approximately 400 trawl samples sweeping a seabed area of approximately 30km$^2$ each year. The whole North Sea covers an area of approximately 570,000km$^2$ and an average ICES rectangle covers an area of 3,500km$^2$. Thus while the sampling statistics appear impressive, and represent a substantial economic investment on the part of the participating countries, in reality the actual fraction of the total North Sea, and its resident individual populations, sampled in any one year is relatively small. Deriving metric values that convey something approaching reality therefore requires 20 or more individual trawl samples to be combined to generate a single representative sample (Greenstreet and Piet, 2008). The consequence of this is that in order to examine temporal trends at the resolution of single years, data collected across ten or more ICES rectangles need to be combined. This clearly limits the spatial resolution at which these surveys can be used to address questions regarding variation in fish biodiversity and food web dynamics. The value of groundfish surveys therefore starts to diminish at spatial scales smaller than the spatial units used in, for example, the UK Charting Progress 2 process. A further consequence of this scale issue is that coastal areas are relatively under-surveyed. This is also partly due to the nature of the vessels generally involved in groundfish surveys, which rarely operate in water shallower than 30m.

Indicator derivation

Firstly, it was necessary to determine which of the indicators stipulated in the EC Decision document (Table 1) could be populated with regard to the fish sub-component. “Habitat” level metrics were considered not appropriate for fish; although there may be a need to ensure that fish-related habitat issues, for example conservation of key habitats such as spawning grounds, were adequately covered in the development of “habitat” indicators. Even here though, problems related to the availability of key fish habitats should be reflected by changes in the “species” level indicators set for criteria 1.1 species distribution, 1.2 population size,
and 1.3 *population condition*. Indicator 1.1.3 *area covered by the species* (for sessile and benthic species) was also considered not relevant to mobile species such as fish; related issues in respect of fish would be addressed via indicator 1.1.1 *distributional range*. Finally, in respect of indicator 1.3.2 *population genetic structure*, whilst there is evidence of population genetic structuring in some fish species, these invariably involve commercially targeted species such as cod (Hutchinson et al., 2001; Nielsen et al., 2009). For the vast majority of fish species there simply are no data. Since genetically separate fish populations would require a degree of spatial segregation, any issues associated with a decline in one or more genetically distinct populations would be reflected by changes in range extent.
Table 1. Levels, criteria and indicator types proposed for Descriptor 1 “Biological diversity is maintained” and Descriptor 4 “Food webs” of the MSFD (EC 2010).

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Criterion 4.1 was also considered to be largely irrelevant to fish indicators and targets. This criterion, and its associated indicator, was considered to relate primarily to “reproductive” productivity, for example kittiwake chick production per breeding pair, which has been shown to be closely linked to the availability of suitable prey (Frederiksen et al., 2004; Daunt et al., 2008). In fish populations, recruit production, the main measure of successful reproductive productivity, is highly variable. Its relationship to spawning stock size is generally weak and rarely statistically significant until spawning stock size falls to critically low levels, at which
point recruit production often declines markedly. This variability in recruit production is the consequence of small variations in the exceptionally high mortality rates that occur during a number of processes between the egg and juvenile phases of the life cycle. Recruit production is therefore considered to be primarily driven by a number of stochastic processes strongly influenced by environmental factors, and generally not related to anthropogenic activities. The possibility of using growth productivity indicators, e.g. Production/Biomass (P/B) ratios, was instead considered. However, for most species, but particularly “sensitive” species, we would want populations to exhibit low P/B ratios; populations dominated by older, larger, mature fish, close to the asymptote of their growth curves, and therefore exhibiting low growth production per unit biomass. However, it was thought that this would be “counter-intuitive” for most stake-holders, and could potentially confuse matters.

Thus, in respect of demersal fish, metrics could be identified, or suggested, that could fulfill indicator roles for: 1.1.1 Distributional range; 1.1.2 Distributional pattern within the range; 1.2.1 Population abundance and biomass; 1.3.1 Population demographic characteristics; 1.7.1 Composition and relative proportions of ecosystem components; 4.2.1 Large fish (by weight); and 4.3.1 Abundance trends of functionally important selected groups/species. However, some of these metrics were considered insufficiently understood at the current time and would therefore require further development before appropriate targets for them could be set.

**Distribution range**

Geographic shifts (i.e., northwards or southwards) in a species’ range were considered primarily to be a response to environmental change, whereas changes in the extent of a species’ range may well be a response to pressure from human activities. Indicators for this criterion also needed to take account of differences in situation; for example, continental shelf communities or shelf-edge communities. Thus for a continental shelf sea, variation in range considered the horizontal plane, where as for shelf-edge seas, variation in range considered the vertical axis (depth). For each species-based “range” indicator \( (R_s) \) therefore, the metric proposed considers the proportion of ICES rectangles (shelf sea) (equation 1), or the proportion of sampled depth bands (shelf-edge sea) (equation 2),

\[
R_{s,y} = \frac{C_{s,y}}{C_{Total,y}}
\]

\[
R_{s,y} = \frac{D_{s,y}}{D_{Total,y}}
\]

sampled by each annual survey in which the species \((s)\) in question is recorded present in each year \((y)\). \(C_{s,y}\) and \(D_{s,y}\) are, respectively, the number of ICES rectangles or depth bands that the species in question is recorded present in each year and \(C_{Total,y}\) and \(D_{Total,y}\) are, respectively, the number of ICES rectangles or depth bands sampled in the survey in each year.

**Distributional pattern within the range**

This indicator was considered to be concerned about variation in the distribution of individuals within the occupied range (ie. \(C_{s,y}\), see equation 2); increases or decreases in the patchiness (contagion) or evenness (dispersion) of individuals over the area occupied. As a simple dispersion/contagion metric, the mean/variance ratio was therefore proposed.

---

\(^5\) Distributional range as suggested here relates to the “extent” of a species’ distribution and not the geographical position of its distribution.
This metric can be computed using either ICES rectangle abundance data (numbers km\(^2\)) or biomass data (kg km\(^2\)), both of which can be substituted in equation 3 as \(A_{s,y,c}\), the abundance or biomass of a given species (s) in each year (y) in each ICES rectangle (c) where it is present, and simply involves calculating the mean of the values across all the rectangles in which a species was recorded present in any one year (numerator part of equation 3) and dividing this by the variance of these values (denominator part of equation 3).

In a Poisson distribution, which describes the distribution of randomly dispersed objects, the mean and the variance are equal; a mean:variance ratio of 1.0 therefore indicates a random distribution of individuals over the occupied area. Values <1.0 (variance>mean) indicate patchy or contagious distributions while values >1.0 (mean>variance) indicate even or dispersed distributions; declining trends reflect increasing contagion, while increasing trends suggest increasing dispersion. Given the generally restricted number of depth bands considered in most studies of shelf-edge fish communities, this approach may not be applicable in this situation.

**Population abundance and/or biomass**

Both abundance and biomass data are readily available from the majority of groundfish surveys, or easily determined through application of weight-at-length relationships to the abundance data. Metrics of both population abundance and biomass are therefore generally available. The question is whether both are necessary in respect of fish for the biodiversity indicator? Some pressures from human activities, such as fishing, are size selective (targeting the larger individuals) and raise mortality rates. Any increase in mortality reduces life expectancy, so average age in a population declines. Being non-deterministic in their growth, any reduction in average age in a fish population results in a reduction in average length. Since weight varies with length as a cubic power, pressures that cause mortality rates to increase affect population biomass to a greater extent than they affect population abundance. On the other hand, other anthropogenic pressures on the marine ecosystem, such as chronic or acute pollution events might influence the productivity of particular populations. Such effects might be better detected using metrics of population abundance. There are therefore good arguments to support the use of indicators of both population abundance and population biomass. Since we have the capacity to provide both, this is the obvious way forward.

The metrics used to act as abundance and biomass indicators in the North Sea pilot study standardise the abundance and biomass data by taking account of the area swept by the trawl gear in each annual survey, thus:
A_{s,y} = \frac{\sum_{c=1}^{C_{\text{Total},y}} A_{s,y,c}}{\sum_{c=1}^{C_{\text{Total},y}} \phi_{s,y,c}} \Phi_{\text{Std}} \quad \text{(4.)}

B_{s,y} = \frac{\sum_{c=1}^{C_{\text{Total},y}} B_{s,y,c}}{\sum_{c=1}^{C_{\text{Total},y}} \phi_{s,y,c}} \Phi_{\text{Std}} \quad \text{(5.)}

A_{s,y}$ and $B_{s,y}$ are the desired annual species-based indicators. These are derived by first summing the abundances $A_{s,y,c}$ (or biomasses $B_{s,y,c}$) of the species in question ($s$) in each year ($y$) in each ICES rectangle ($c$) surveyed across all the rectangles surveyed in each year ($C_{\text{Total},y}$). Next the total area of seabed swept in each annual survey is determined by summing the areas swept in each ICES rectangle ($\phi_{y,c}$) across all rectangles surveyed in each year and the abundance (or biomass) sum is divided by this value to derive “whole survey” density estimates (numbers km$^{-2}$ or biomass kg km$^{-2}$ at a “regional” scale (in MSFD terms “sub-regional” or “sub-sub-regional” scale), i.e. the area covered by the survey (in our pilot study, the North Sea covered by the ICES Q1 IBTS). We have used are square kilometres (km$^2$) because this seems a sensible spatial unit by which to consider most fish populations. However, many of the rarest fish species will be recorded at densities of <1km$^{-2}$ (or <1kg km$^{-2}$), and the combination of values <1 and >1 can cause analytical problems, particularly if transformation is required. Consequently, these survey density estimates were raised by a “standard” area ($\Phi_{\text{Std}}$), which in the case of the North Sea pilot study was 30km$^2$, this being an approximation of the mean or median area surveyed across all years of the ICES Q1 IBTS.

If species biomass in any year and rectangle are not available directly from the survey, this can be estimated from the species abundance at length data (per year and rectangle) by applying species specific weight-at-length relationships of the form $W_{s,l} = \alpha_l^{\beta_l}$, where $\alpha_l$ and $\beta_l$ are, respectively, the species-specific constant and exponent parameters of the power function, $l$ is the length in question and $W_{s,l}$ is the corresponding weight of an individual fish of this species and length. Biomass estimates for each species in each year and rectangle ($B_{s,y,c}$) can then be determined as $B_{s,y,c} = \sum_{l=\text{Min}}^{l=\text{Max}} W_{s,l,y,c} A_{s,l,y,c}$, where $W_{s,l,y,c}$ is the weight of each individual fish of species $s$ and length $l$ in year $y$ rectangle $c$ and $A_{s,l,y,c}$ is their abundance.

**Population demographic characteristics**

Data to assess variation in fish population demographics are not routinely collected as part of the groundfish survey protocols. However, it is possible to generate some potential metrics using life-history trait information and applying these to the standard abundance-at-length data provided by the surveys. Thus length at first maturity data are available for many species (see Table 2), and where they are not, they can be estimated from von Bertalanffy growth equation ultimate body length parameter values (see Table 2) using regression analysis (Figure 1). In any year, and for any species, sampled in a survey, the proportion of individuals, or the proportion of the sampled biomass, that exceeds this length can be...
determined. This provides an estimate of the proportion of mature individuals ($M_{Nos,s,y}$) or biomass ($M_{Biom,s,y}$) in the population\(^6\). The indicator is calculated as:

$$
M_{Nos,s,y} = \frac{\sum_{c=1}^{C_{Nos,s,y}} \sum_{l=1}^{L_{Nos,s,y}} A_{l,y,c}}{\sum_{c=1}^{C_{Nos,s,y}} A_{l,y,c}}
$$

$$
M_{Biom,s,y} = \frac{\sum_{c=1}^{C_{Biom,s,y}} \sum_{l=1}^{L_{Biom,s,y}} B_{l,y,c}}{\sum_{c=1}^{C_{Biom,s,y}} B_{l,y,c}}
$$

\(^6\) It is important to realise that variation in this metric as defined actually reflects variation in the size composition of the population, rather than any change over time in the length at which individuals of a particular species mature.

Figure 1. Relationship between Length at maturity ($L_{\text{max}}$) and the von Bertalanffy ultimate body length ($L_\infty$) parameter for 60 demersal species where estimates for both parameters were available. Each species data point is identified by the MS 3-letter species code.

Composition and relative proportions of ecosystem components

This indicator was interpreted as the species composition and relative proportions of species in the demersal fish community. Univariate community metrics that have commonly been applied to fish communities could therefore be used for the purposes of this indicator (Greenstreet and Hall, 1996; Greenstreet et al., 1999; Rogers et al., 1999; Piet and Jennings 2005; Greenstreet and Rogers, 2006). In respect specifically of biodiversity, these include metrics of species evenness (Pielou’s evenness index, the Shannon-Weiner index, Simpson’s index, and Hill’s N1 and N2, which are the exponential and reciprocal of the last two metrics respectively) and metrics of species richness (species counts or Margaleff’s richness) (Greenstreet et al., in press). However, it is not clear what targets might be set for these...
metrics. The Q1 IBTS case study presented here has demonstrated that both species richness (Figure 1) and species evenness (Figure 28) have increased over the duration of the survey time-series.

The current OSPAR EcoQO for “fish communities” is based on the large fish indicator (LFI) (the proportion by weight of fish in the community larger than a specified length threshold). The LFI was specifically developed to monitor the effect of fishing pressure on the status of the broader fish community, beyond just the commercial stocks that are the subject of annual assessments. Under the auspices of ICES, the LFI has been the focus of a considerable research and development effort, with attention paid to: refining its sensitivity to fishing pressure; determining appropriate targets; developing an appropriate theoretical modelling framework to underpin management advice; and “rolling the process out” to redefine the LFI and set appropriate targets for additional marine regions beyond just the North Sea pilot area (Greenstreet et al., 2011; Shephard et al., in press). The LFI is therefore a prime candidate as an indicator of change in the composition and relative proportions of ecosystem [fish community] components. However, the LFI is also explicitly mentioned in the EC 2010 Decision Document as the indicator for Criterion 4.2 proportion of selected species at the top of food webs (see next section), but there seems to be no prohibition of the same indicator fulfilling two different indicator roles across different Descriptors.

Large fish (by weight)
The use of the large fish indicator (LFI) in a food web context is firmly rooted in size-based aquatic food web theory (Kerr & Dickie, 2001). Body-size is widely regarded as having at least as important a role to play in the processes structuring marine communities and controlling food web dynamics as species identity (Jennings et al., 2001; 2002; Jennings & Mackinson 2003). The LFI was developed as part of the OSPAR EcoQO pilot study carried out for the North Sea (Greenstreet et al., 2011). However, the indicator is region and survey specific. It is the underlying process used in the North Sea to define and set targets for the indicator (Greenstreet et al., 2011) that needs to be "rolled out" to other regions and surveys, rather than simply applying the North Sea indicator definition and targets in a "prescribed" fashion (Shephard et al., in press). Derivation of the indicator is described in detail in the two cited studies, but in essence the LFI is defined as:

\[
LFI = \frac{\sum_{c=1}^{C_{\text{tax}}}{\sum_{x=1}^{S}{\sum_{l=1}^{l_{\text{LFI, threshold}}}{B_{x,y,l}c}}}}{\sum_{x=1}^{S}{B_{x,y}}}
\]

where S is the total number of species in the sample and \(l_{\text{LFI, Threshold}}\) is the region/survey specific large fish length threshold (see equation 5 and associated text for explanation of the remaining terms) and \(l_{x,\text{max}}\) is the maximum length of each species recorded in the survey.

Abundance trends of functionally important selected groups/species
Fish trophic groups have been defined in previous studies. For example, the European Regional Seas Ecosystem Model (ERSEM) defines and models carbon flow through four fish groups: pelagic planktivore; pelagic piscivores; demersal benthivores; and demersal piscivores (Greenstreet et al., 1997). For ERSEM, species were assigned to each group on the basis of the adult fish diet. However, fish diet varies markedly with age and length (Hislop, 1997; Greenstreet, 1996; Greenstreet et al., 1998), and more meaningful indicators might be developed by taking account of both species- and size-related variation in diet.
Selection of species-specific indicators

The wealth of data that groundfish surveys provide in itself also poses problems: for example in deciding the basis on which particular species are selected to fulfil specific indicator roles. The Q1 IBTS analysed here has, over the course of its 26y history up to 2008, generated data on 128 demersal fish species alone. Our analyses have illustrated the variety of different ways that particular aspects of these species have varied over this time. Simply assessing variation in 128 different metrics fulfilling each of the different indicator roles listed in Table 1 would present considerable difficulties, particularly with regard to target setting. In order to interpret trends in indicator values for particular species, and then set targets for these indicators, it is necessary to have some a priori expectation as to what the metric value might be under conditions of sustainable exploitation; most particularly, what this value might be compared with present day or recent historic values.

The sensitivity of different fish species to human pressure has been linked to their life-history characteristics. Species with \( k \)-type traits, such as large ultimate body size, slow growth rate, low fecundity, late age and large size at maturity, etc, are generally more sensitive to human activities that increase mortality rates (Jennings et al., 1998; Jennings et al., 1999; Gislason et al., 2008). Human pressure on marine ecosystems generally brings about a decline in such species (Jennings et al., 1998; Jennings et al., 1999), for example the well documented decline in many elasmobranch species in the North Sea (Walker & Heessen, 1996; Walker & Hislop, 1998; Greenstreet & Rogers, 2000). Conversely, \( r \)-type species, which have the opposite traits, are opportunistic in nature and tend to flourish in disturbed situations, or when populations of their \( k \)-type predators are diminished (Greenstreet et al. 1999; Daan et al., 2005; Greenstreet et al. 2011). Opportunistic species often respond positively to other pressures on the ecosystem, such as chronic enrichment pollution. Taking account of species life-history characteristics can therefore provide the basis for selecting particular species to perform the indicator roles set out in the EC 2010 Indicator Decision document for Descriptor 1 (Table 1).

A further requirement for species selected to perform an indicator role is that they are adequately represented in the groundfish survey time-series. In the case study presented here, representation in at least half of the years that the Q1 IBTS was carried out was deemed a key requirement and 76 species met this condition. These 76 species were then ranked according to three life-history traits, ultimate body length (\( L_{\infty} \)), von Bertalanffy growth parameter (\( K \)), and length at first maturity (\( L_{\text{maturity}} \)). The average of these three rankings was determined, and in turn ranked to place each species along an \( r \)-\( k \) spectrum with values from 1 (most opportunistic species) to 76 (most sensitive species) (Figure 2A). 33\%iles then defined three groups of species: the 25 lowest ranked species, considered to be \( r \)-type or opportunist species; the 25 highest ranked species, considered to be \( k \)-type sensitive species, and a group of 26 middle ranked \( r/k \)-type species considered to be intermediate in their opportunistic/sensitivity traits (Table 2). Variation in the three life-history traits across these three groups is illustrated in the Box-Whisker plots shown in Figure 2B. Overlap in either of the three traits was minimal between the two extreme groups; sensitive \( k \)-type species and opportunist \( r \)-type species\(^7\).

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\(^7\) Figure 2 and Table 2 define the sensitive \( k \)-type and opportunist \( r \)-type species groups based on the North Sea Q1 IBTS demersal fish case study. These groups would need to be defined explicitly and specifically for each survey data set used in any given marine region assessment. Further note that this species grouping, based on species-specific life-history traits, is particularly appropriate in respect of pressures that increase rates of mortality above
Figure 2. A. Variation in ultimate body length \( (L_{\text{infinity}}) \) von Bertalanffy growth parameter \( (K) \), and length at first maturity \( (L_{\text{maturity}}) \) with increasing ranking on the \( r-k \) scale. Dashed vertical lines indicate the upper, middle and lower data 33%iles, representing sensitive, intermediate and opportunist species. B. Box-whisker plots showing the median, upper and lower quartiles, and maximum and minimum ultimate body length \( (L_{\text{infinity}}) \) von Bertalanffy growth parameter \( (K) \), and length at first maturity \( (L_{\text{maturity}}) \) values for three groups of species, opportunists \( (r) \), sensitive \( (k) \) and intermediate trait species \( (r/k) \).

If a particular anthropogenic pressure affects fish species in other ways, some alternative approach to identifying “sensitive” species may be necessary.
Table 2. List of 76 species recorded present in the Q1 IBTS in at least half (13y or more) of the survey's time-series giving their r-k spectrum ranking and group, and detailing their life-history trait data on which this ranking and grouping was based. Shaded cells indicate details for species currently listed on the OSPAR list of threatened and endangered species.

<table>
<thead>
<tr>
<th>r’k group</th>
<th>Scientific name</th>
<th>Common name</th>
<th>Ultimate body length (cm)</th>
<th>Von Bertalannfy growth parameter</th>
<th>Length at first maturity (cm)</th>
<th>r-k spectrum rank</th>
</tr>
</thead>
<tbody>
<tr>
<td>Opportunist</td>
<td>Aphia minuta</td>
<td>Transparent goby</td>
<td>5.40</td>
<td>2.230</td>
<td>3.53</td>
<td>1</td>
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<td></td>
<td>Pomatoschistus minutus</td>
<td>Sand goby</td>
<td>9.20</td>
<td>0.928</td>
<td>5.74</td>
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<td>Callionymus reticulatus</td>
<td>Reticulated dragonet</td>
<td>8.76</td>
<td>0.495</td>
<td>5.49</td>
<td>3</td>
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<tr>
<td></td>
<td>Liparis montagui</td>
<td>Montagu’s sea snail</td>
<td>9.54</td>
<td>0.472</td>
<td>5.93</td>
<td>4</td>
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<tr>
<td></td>
<td>Buglossidium luteum</td>
<td>Solenette</td>
<td>11.70</td>
<td>0.540</td>
<td>7.00</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Phrynorhombus norvegicus</td>
<td>Norwegian topknot</td>
<td>9.54</td>
<td>0.472</td>
<td>5.93</td>
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<td>Arnoglossus laterna</td>
<td>Scaldfish</td>
<td>15.80</td>
<td>0.840</td>
<td>9.40</td>
<td>7</td>
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<td>Liparis liparis</td>
<td>Sea snail</td>
<td>11.87</td>
<td>0.417</td>
<td>7.24</td>
<td>8</td>
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<td></td>
<td>Echichthys vipera</td>
<td>Lesser weever</td>
<td>11.87</td>
<td>0.417</td>
<td>7.24</td>
<td>9</td>
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<td></td>
<td>Callionymus maculatus</td>
<td>Spotted dragonet</td>
<td>12.65</td>
<td>0.402</td>
<td>7.67</td>
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<td>Agonus cataphractus</td>
<td>Hooknose</td>
<td>17.40</td>
<td>0.419</td>
<td>9.00</td>
<td>11</td>
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<td>Syngnathus rostellatus</td>
<td>Nilsson's pipefish</td>
<td>20.00</td>
<td>0.747</td>
<td>11.65</td>
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<td>Lycenchelys sarsii</td>
<td>Sar's wolf eel</td>
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<td>9.36</td>
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<td>Trisopterus minutus</td>
<td>Poor cod</td>
<td>20.30</td>
<td>0.506</td>
<td>13.00</td>
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<td>Triglops murray</td>
<td>Moustache sculpin</td>
<td>15.73</td>
<td>0.355</td>
<td>9.36</td>
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<td>Callionymus lyra</td>
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<td>Ciliata mustela</td>
<td>Five-bearded rockling</td>
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<td>Norway pout</td>
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<td>0.520</td>
<td>19.00</td>
<td>18</td>
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<td>Topknot</td>
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<td>0.314</td>
<td>11.43</td>
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<td>Sea scorpion</td>
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<td>Microchirus variegatus</td>
<td>Thickback sole</td>
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<td>Pholis gunnellus</td>
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<td>Raniceps raninus</td>
<td>Tadpole fish</td>
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<td>12.64</td>
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<td>Mullus surmuletus</td>
<td>Striped red mullet</td>
<td>33.40</td>
<td>0.430</td>
<td>18.61</td>
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<tr>
<td>$r/k$ group</td>
<td>Scientific name</td>
<td>Common name</td>
<td>Ultimate body length (cm)</td>
<td>Von Bertalanffy growth parameter</td>
<td>Length at first maturity (cm)</td>
<td>$r$-$k$ spectrum rank</td>
</tr>
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</tr>
<tr>
<td></td>
<td><em>Hippoglossoides platessoides</em></td>
<td>Long rough dab</td>
<td>24.60</td>
<td>0.336</td>
<td>15.00</td>
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<td><em>Capros aper</em></td>
<td>Boarfish</td>
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<td>0.283</td>
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<td><em>Limanda limanda</em></td>
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<td><em>Myoxocephalus scorpius</em></td>
<td>Bullrout</td>
<td>34.00</td>
<td>0.240</td>
<td>15.00</td>
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<td><em>Chelidonichthys cuculus</em></td>
<td>Red gurnard</td>
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<td>0.460</td>
<td>25.00</td>
<td>29</td>
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<td><em>Merlangius merlangus</em></td>
<td>Whiting</td>
<td>42.40</td>
<td>0.320</td>
<td>20.00</td>
<td>30</td>
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<td><em>Microstomus kitt</em></td>
<td>Lemon sole</td>
<td>37.10</td>
<td>0.415</td>
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<td>31</td>
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<td><em>Enchelyopus cimbrius</em></td>
<td>Four-bearded rockling</td>
<td>35.90</td>
<td>0.196</td>
<td>14.00</td>
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<td><em>Solea vulgaris</em></td>
<td>Dover sole</td>
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<td><em>Syngnathus acus</em></td>
<td>Great pipefish</td>
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<td><em>Trisopterus luscus</em></td>
<td>Bib</td>
<td>38.00</td>
<td>0.211</td>
<td>23.00</td>
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</tr>
<tr>
<td>Intermediate</td>
<td><em>Lycodes vahlii</em></td>
<td>Vahl's eelpout</td>
<td>40.09</td>
<td>0.209</td>
<td>21.98</td>
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<td>Snake blenny</td>
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<td>0.165</td>
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<td>40</td>
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<td>Snake pipefish</td>
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<tr>
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<td><em>Scophthalmus rhombus</em></td>
<td>Brill</td>
<td>50.00</td>
<td>0.270</td>
<td>37.00</td>
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<tr>
<td></td>
<td><em>Eutrigla gurnardus</em></td>
<td>Grey gurnard</td>
<td>46.16</td>
<td>0.156</td>
<td>21.00</td>
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<tr>
<td></td>
<td><em>Psetta maxima</em></td>
<td>Turbot</td>
<td>57.00</td>
<td>0.320</td>
<td>46.00</td>
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<tr>
<td></td>
<td><em>Sebastes viviparus</em></td>
<td>Norway haddock</td>
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<td>0.070</td>
<td>20.90</td>
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<tr>
<td></td>
<td><em>Gaidropsarus vulgaris</em></td>
<td>Three-bearded rockling</td>
<td>47.50</td>
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<td>27.00</td>
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</tr>
<tr>
<td></td>
<td><em>Helicolenus dactylopterus</em></td>
<td>Bluemouth</td>
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<td>0.095</td>
<td>25.00</td>
<td>47</td>
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<tr>
<td></td>
<td><em>Lepidorhombus whiffiangon</em></td>
<td>Megrim</td>
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<td>0.073</td>
<td>19.00</td>
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<tr>
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<td>0.233</td>
<td>46.00</td>
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<tr>
<td></td>
<td><em>Myxine glutinosa</em></td>
<td>Hagfish</td>
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<td>0.164</td>
<td>25.00</td>
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<tr>
<td></td>
<td><em>Melanogrammus aeglelinus</em></td>
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<td>68.30</td>
<td>0.190</td>
<td>34.00</td>
<td>51</td>
</tr>
<tr>
<td>r/k group</td>
<td>Scientific name</td>
<td>Common name</td>
<td>Ultimate body length (cm)</td>
<td>Von Bertalanffy growth parameter</td>
<td>Length at first maturity (cm)</td>
<td>r-k spectrum rank</td>
</tr>
<tr>
<td>----------</td>
<td>--------------------------</td>
<td>---------------------</td>
<td>---------------------------</td>
<td>---------------------------------</td>
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<td>------------------</td>
</tr>
<tr>
<td>Sensitive</td>
<td>Zoarces viviparus</td>
<td>Viviparous blenny</td>
<td>52.00</td>
<td>0.130</td>
<td>27.86</td>
<td>52</td>
</tr>
<tr>
<td></td>
<td>Cyclopterus lumpus</td>
<td>Lumpsucker</td>
<td>55.00</td>
<td>0.120</td>
<td>29.33</td>
<td>53</td>
</tr>
<tr>
<td></td>
<td>Pleuronectes platessa</td>
<td>Plaice</td>
<td>54.40</td>
<td>0.110</td>
<td>27.00</td>
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<tr>
<td></td>
<td>Chelidonichthys lucerna</td>
<td>Tub gurnard</td>
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<td>0.148</td>
<td>34.15</td>
<td>55</td>
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<tr>
<td></td>
<td>Pollachius pollachius</td>
<td>Pollack</td>
<td>85.60</td>
<td>0.186</td>
<td>43.91</td>
<td>56</td>
</tr>
<tr>
<td></td>
<td>Scyliorhinus canicula</td>
<td>Lesser spotted dogfish</td>
<td>90.00</td>
<td>0.200</td>
<td>58.00</td>
<td>57</td>
</tr>
<tr>
<td></td>
<td>Raja clavata</td>
<td>Thornback ray</td>
<td>105.00</td>
<td>0.220</td>
<td>65.00</td>
<td>58</td>
</tr>
<tr>
<td></td>
<td>Gadus morhua</td>
<td>Cod</td>
<td>123.10</td>
<td>0.230</td>
<td>70.00</td>
<td>59</td>
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<tr>
<td></td>
<td>Anguilla anguilla</td>
<td>European eel</td>
<td>83.20</td>
<td>0.076</td>
<td>42.78</td>
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</tr>
<tr>
<td></td>
<td>Squalus acanthias</td>
<td>Spurdog</td>
<td>90.20</td>
<td>0.150</td>
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<td>61</td>
</tr>
<tr>
<td></td>
<td>Merluccius merluccius</td>
<td>Hake</td>
<td>103.60</td>
<td>0.107</td>
<td>37.00</td>
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</tr>
<tr>
<td></td>
<td>Mustelus asterias</td>
<td>Starry smooth hound</td>
<td>105.71</td>
<td>0.121</td>
<td>53.23</td>
<td>63</td>
</tr>
<tr>
<td></td>
<td>Brosme brosme</td>
<td>Torsk</td>
<td>88.60</td>
<td>0.080</td>
<td>50.00</td>
<td>64</td>
</tr>
<tr>
<td></td>
<td>Raja montagui</td>
<td>Spotted ray</td>
<td>97.80</td>
<td>0.148</td>
<td>67.00</td>
<td>65</td>
</tr>
<tr>
<td></td>
<td>Lophius piscatorius</td>
<td>Angler</td>
<td>135.00</td>
<td>0.176</td>
<td>75.00</td>
<td>66</td>
</tr>
<tr>
<td></td>
<td>Leucoraja naevus</td>
<td>Cuckoo ray</td>
<td>91.64</td>
<td>0.109</td>
<td>59.00</td>
<td>67</td>
</tr>
<tr>
<td></td>
<td>Chimaera monstrosa</td>
<td>Rabbit ratfish</td>
<td>113.10</td>
<td>0.116</td>
<td>56.61</td>
<td>68</td>
</tr>
<tr>
<td></td>
<td>Galeorhinus galeus</td>
<td>Tope</td>
<td>163.00</td>
<td>0.168</td>
<td>120.00</td>
<td>69</td>
</tr>
<tr>
<td></td>
<td>Anarhichas lupus</td>
<td>Catfish</td>
<td>117.40</td>
<td>0.047</td>
<td>43.00</td>
<td>70</td>
</tr>
<tr>
<td></td>
<td>Raja brachyura</td>
<td>Blond ray</td>
<td>139.00</td>
<td>0.120</td>
<td>100.00</td>
<td>71</td>
</tr>
<tr>
<td></td>
<td>Pollachius virens</td>
<td>Saithe</td>
<td>177.10</td>
<td>0.070</td>
<td>55.00</td>
<td>72</td>
</tr>
<tr>
<td></td>
<td>Molva molva</td>
<td>Ling</td>
<td>183.00</td>
<td>0.118</td>
<td>85.00</td>
<td>73</td>
</tr>
<tr>
<td></td>
<td>Hippoglossus hippoglossus</td>
<td>Halibut</td>
<td>204.00</td>
<td>0.100</td>
<td>83.00</td>
<td>74</td>
</tr>
<tr>
<td></td>
<td>Mustelus mustelus</td>
<td>Smooth hound</td>
<td>205.00</td>
<td>0.060</td>
<td>97.39</td>
<td>75</td>
</tr>
<tr>
<td></td>
<td>Dipturus batis</td>
<td>Skate</td>
<td>253.70</td>
<td>0.057</td>
<td>155.00</td>
<td>76</td>
</tr>
</tbody>
</table>
Commercially exploited species used as part of the assessment for Descriptor 3 were deemed eligible for inclusion as indicator species for Descriptors 1 and 4 (provided they met the selection criteria set out above) because they generally constitute key members of the demersal fish community. Where Descriptor 3 species contributed to the breaching of targets set for individual descriptor 1 or 4 indicators and criteria, this should be taken as an indication that management of that species through the CFP was either inadequate or not being applied appropriately.

**Target setting**

When the status of the demersal fish community is deemed to be unsatisfactory, i.e., below good environmental status (GES), targets for $k$-type species should reflect the need to encourage an increase in their abundance. If fish community status is considered adequate (i.e., at GES), then targets should reflect the need to ensure that populations of $k$-type species are maintained and do not decline as a consequence of human activity. Conversely, $r$-types species may well be over-represented in perturbed communities, and even when fish community status might be considered adequate, one would not wish to see any undue increase in their abundance. Targets for $r$-type should at all times therefore reflect the need to ensure that, generally, populations of these opportunist species do not increase.

Targets for opportunist $r$-type species were considered necessary because:

- Such species may be more responsive to human pressures other than fishing, such as pollution enrichment events, which may affect local productivity.
- By the very nature of their life-histories, such species may respond more quickly to both natural and anthropogenic disturbance, thereby potentially providing “early-warning” of impending issues.
- Opportunist species are often amongst the most abundant species in fish communities. They therefore have the potential, if their populations increase, to exert high predation pressure on their prey, which could include the eggs and larval phases of sensitive $k$-type species. Similarly, juveniles of sensitive $k$-type species, as they grow through their small size phase, could be subjected to higher levels of competition from opportunist $r$-type species.

For several of the indicators described above, no targets have been proposed. In these cases, the relationship between the proposed indicator and anthropogenic pressure on the fish community was not considered to be sufficiently well understood as to allow reliable targets to be proposed. The species-based indicators falling into this category were 1.1.2 distributional pattern within the range and 1.3.1 population demographic characteristics. Targets could be proposed for 1.1.1 distributional range, 1.2.1 population abundance and biomass, 1.7.1 composition and relative proportions of ecosystem components, and 4.2.1 large fish (by weight).

To date, most fish-focused marine ecosystem-related work (i.e., not stock assessment science) has focused on setting reference points (or baselines) and targets for community level univariate indicators (Greenstreet and Rogers, 2006); e.g., the 1983 reference period that gave rise to a target of 0.3 for the North Sea demersal fish community LFI (Greenstreet et al., 2011). The rationale underlying this approach does not apply to the single species based indicators stipulated by the EC 2010 Decision document for “species” level criteria (1.1, 1.2, and 1.3). Reference periods for different species differ markedly depending on each species’ life-history characteristics. Because species ranked higher along the $r$-$k$ spectrum have lower tolerance to pressure from human activities, they would have been adversely affected by activities such as fishing earlier than lower ranked species. For many $k$-type species, notable fishing impacts would already have occurred long before the advent of the Q1 IBTS. The current proposal therefore uses the entire groundfish survey time series as a “reference” and compares indicator values in the latest assessments with the mean and standard deviation for all data over the full survey time series. Essentially targets would be set based on the recent deviation in metric values compared
with variation over period covered by each survey. Target deviation direction and extent would then depend on whether the species in question was considered to be a sensitive \( k \)-type or opportunist \( r \)-type species, and whether the management situation involved conservation of a situation currently at GES, or restoration of a community towards GES. For those indicators considered sufficiently well understood as to allow targets to be proposed, a probabilistic approach to target setting was developed with the objective of being able to assess whether criterion-level targets were achieved with a specified level of statistical significance.

**Analytical procedures**

Factor analysis performed on each of the species based indicators suggested between 5 and 8 general types of temporal trend (see accompanying North Sea case study). For the distributional pattern indicators, basing the mean/variance ratio on either ICES rectangle abundance data or biomass data, seven or eight trend types respectively were identified, none of which were monotonic in nature. For each of the remaining indicators, only one of the factor score trends was monotonic, and just 24% to 55% (average 44%) of the 49 species analysed were linked to the monotonic factor (only 49 species, those recorded present in all years of the ICES Q1 IBTS, were analysed). For all six indicators examined therefore, the use of simple trend based statistics to set targets would have been questionable.

An alternative approach is therefore proposed. This essentially uses the entire time series of each indicator to provide the “reference level” or “baseline”. Firstly, for each species-based \( s \) indicator time series, the mean indicator value \( \bar{I}_s \) and the standard deviation around this mean \( \sigma_I \) across all years of available data in the indicator time series \( Y_s \) (from the first \([y_{\text{first}}]\) to the last \([y_{\text{last}}]\)) was calculated.

\[
\bar{I}_s = \frac{\sum_{y=1}^{y_{\text{last}}} I_{s,y}}{y_{\text{last}} - y_{\text{first}}}
\]

\[
\sigma_I = \sqrt{\frac{\sum_{y=1}^{y_{\text{last}}} (I_{s,y} - \bar{I}_s)^2}{y_{\text{last}} - y_{\text{first}}}}
\]

where \( I_{s,y} \) is the indicator value for a given species and year. Each year's indicator value can then be converted to its standardised deviate \( I_{s,y,\text{STD.DEV}} \) equivalent by

\[
I_{s,y,\text{STD.DEV}} = \frac{I_{s,y} - \bar{I}_s}{\sigma_I}
\]

Abundance and biomass data are notoriously non-normal in their distribution, generally being skewed to the left (long “tail” of a few higher than expected values). Log transformation mitigates this situation and stabilises variances (reduces the tendency for variance to increase with the mean). Before substituting the \( A_{s,y} \) and \( B_{s,y} \) into \( I_{s,y} \) in equations 8, 9, and 10 therefore, the raw indicator values must first be log-transformed.

Calculating the standardised deviates of an indicator’s values does not in any way alter the trend shape. It simply places each indicator trend at the same central point (the mean) and fixes the range in values over a constant scale. This is illustrated using the biomass indicator \( B_{s,y} \) for three sensitive species: angler, spurdog and spotted ray (Figure 3). By equally scaling the variation in each species-specific indicator in this way, each species’ trend can be compared directly with any other species. Between species differences in indicator values (comparisons between high abundance species and low abundance species for example) are eliminated so that
each indicator has equal weighting in terms of their contribution to criterion-level targets. It also means that all indicator-level and criterion-level targets can be expressed using a single term, which is applicable to all species-specific indicators regardless of actual indicator values; greatly simplifying target setting.

Figure 3. Trends in angler fish, spurdog and spotted ray log-transformed biomass indicator values (A) and standardised deviates of the log-transformed indicator values (B) derived from the North Sea Quarter 1 International Bottom Trawl Survey.

Figure 4A shows fitted normal distributions for each of the biomass trends illustrated in Figure 3A; each distribution is determined by the mean and standard deviation of the indicator values. Since all three of these species are sensitive species, if not currently at GES, then our targets would require their biomass to increase in order to achieve GES. Appropriate targets might therefore be “the most recent biomass estimate in each species time series should exceed +1 standard deviation above the long-term mean value”. This would necessitate setting separate individual targets for each species-specific (log-transformed) biomass indicator; for example, 5.48 for Angler fish, 5.29 for spurdog and 4.80 for spotted ray (\(x + \sigma\)); and the same might need to be done for a possible further 73 species in the North Sea case study. This same process would also have to be repeated for each of the other species-specific indicators that are sufficiently well developed as to allow target setting (abundance and distributional range). Furthermore, with each new year of monitoring, with new “recent” indicator values and the each time series extended by another year, this entire target setting procedure would have to be undertaken again. However, converting the basic indicator values to their standardised deviates overlays each distribution, placing each on an identically scaled abscissa (Figure 4B). Indicator targets can now be phrased as “recent indicator standardised deviates should exceed +1”. The same target is relevant to each of the sensitive species, and it could well apply to several or all of the different species-specific indicators. Furthermore, the target remains the same in each and every year till GES is finally achieved.
The biggest advantage to this approach, however, lies in the fact that the normal distribution probability density function can be used to determine the probability of any specified deviation away from the mean (Figure 5). Thus if the target suggested above, “recent indicator standardised deviates should exceed +1”, were to be set for sensitive species, then for every one of the sensitive species monitored (25 in the instance of the North Sea case study) there would be a 16% probability of observing such a value. Given this relatively high probability of observing such a value simply by chance, such a target may initially appear unambitious. However, sensitive species have $k$-type population dynamics. Populations of many of these species may have been reduced to relatively low levels by detrimental human activities and, by their very nature, such species will only be capable of relatively low rates of population recovery. In this context, such a target may now seem biologically unrealistic, particularly if time scales to achieve targets are short. Consequently, more appropriate individual indicator-level targets for sensitive $k$-type species might be “the most recent indicator standardised deviates should exceed +0.5”. The probability of observing such a value, based on each species historic indicator time series, would now be 35%.

Figure 4. A: Normal distributions fitted to the angler fish, spurdog and spotted ray log-transformed biomass indicator values shown in Figure 31A. The mean and standard deviation parameter values for each distribution are shown. B: The same normal distributions applied to the standardised deviates of the log-transformed biomass indicator values.
Any assessment of the state of the fish community would not rely on whether individual species had met their indicator-level targets or not. Instead the assessment would rely on meeting criterion-level targets. Since we know the probability of an individual indicator meeting its target, and we know the number of individual species-specific indicators we have assessed (25 sensitive k-type species in the case of the North Sea demersal fish case study), we can use the binomial distribution to determine the probability of a specified number (or proportion) of these indicators meeting their targets with a pre-determined probability of this number doing so simply by chance. Science generally adopts a 5% probability significance level, and this would seem to be sufficiently ambitious as the basis for setting criterion-level targets. The binomial experiment for the North Sea case study can therefore be summarised as:

- “Assuming individual indicator targets of “the most recent indicator standardised deviate should exceed +0.5”, there is a 35% probability of this happening by chance in respect of each individual indicator. Out of 25 indicators, what number would have to meet this target for there to be a less than 5% probability of this overall result occurring by chance?”

One would expect at least 8 to 9 species to meet their individual targets purely by chance (.35 x 25), but it turns out that if 14 achieve their target, there is a <5% probability that such a result could come about purely by chance. This represents 56% of the 25 sensitive k-type species assessed in the North Sea case study. However, the proportion of indicators that need to meet
their targets is sample size dependent; if only 10 sensitive species had been monitored then 70% would have needed to meet the target, but this proportion drops to 50% if 40 species-specific indicators are analysed (Table 3).

More ambitious targets could be set for the individual species-specific targets; for example the other target previously considered, “the most recent indicator standardised deviate should exceed +1.0”. Now the probability of any individual indicator meeting its target is reduced to 16%. Assuming the same level of significance for meeting the criterion-level target, only 8 of the 25 North Sea sensitive species would have to meet this more ambitious indicator-level target for this to occur by chance with a probability of <5% (Table 3). The combinations of indicator-level and criterion-level targets presented in Table 3 all represent the same level of statistical significance at the criterion-level. This illustrates that the binomial distribution can be used to “trade off” levels of ambition at the indicator- and criterion-levels so as to maintain a constant level of ambition at the criterion-level.

<table>
<thead>
<tr>
<th>Number of individual species indicators</th>
<th>Targets for individual indicators: The most recent indicator standardised deviates are $&gt;1.0$ relative to the long-term mean.</th>
<th>Targets for individual indicators: The most recent indicator standardised deviates are $&gt;0.5$ relative to the long-term mean.</th>
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<tr>
<td>Criterion Target</td>
<td>Proportion</td>
<td>Criterion Target</td>
</tr>
<tr>
<td>------------------</td>
<td>------------</td>
<td>------------------</td>
</tr>
<tr>
<td>10</td>
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<td>0.50</td>
</tr>
<tr>
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<td>0.29</td>
</tr>
<tr>
<td>40</td>
<td>11</td>
<td>0.28</td>
</tr>
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</table>

Table 3: The number of individual species-specific indicators that need to meet their stipulated indicator-level targets in order to achieve the criterion-level target, implying that at the <5% probability significance level, a significant fraction of the individual indicator targets have been met. The table shows how variation in the number of species assessed affects the criterion level targets, and how modifying the level of ambition in the individual species targets affects the fraction of the assessed species that need to meet their indicator targets in order to demonstrate a significant improvement at the criterion level. Green shading shows the situation for the demersal fish analysed in the North Sea case study described in the working document.

So far we have considered the rationale underpinning the setting of criterion-level targets for sensitive $k$-type species when not at GES, and needing improvement in their status at the criterion-level to achieve GES. But once GES is achieved, all that is then required is that current status is conserved; ie no significant decline in the criterion-level status of sensitive species. Thus individual indicator-level targets could be “the most recent indicator standardised deviate should equal or exceed the long-term time series mean (standardised deviate $\geq 0.0$)”. The criterion level target would then be “the proportion of species meeting their individual indicator targets should be sufficiently high that there is a <5% probability of this happening by chance”. Since there is a 50% probability of species meeting their individual indicator-level targets by chance, out of 25 species-specific indicators, between 12 and 13 might be expected to meet their targets through chance alone. But if this was the situation, then that would mean that there was no management in place to actively prevent species failing their targets, or that such management was ineffective. To address this concern, the required binomial experiment can be stated “if there is a 50:50 chance of species meeting their target, then with a given number of species-specific indicators, what number should equal or exceed the long-term time series mean for this to occur by chance with a
probability of <5%"? In the North Sea case study, with 25 sensitive \( k \)-type species, this number turns out to be 18. Thus if 8 species in the North Sea were to fail their target, this could infer that there was a significant risk of impending failure to maintain sensitive demersal fish GES in the North Sea. More than 12 species failing the criterion-level target would be indicative of current actual failure. Table 4 reveals that meeting such "conservation-orientated" criterion-level targets is increasingly difficult the lower the number of individual species-specific indicators assessed. Essentially, the objective of this target is to prove "no change" and "no change" is generally regarded as being the "null hypothesis". Null hypotheses are notoriously difficult to prove, depending heavily on the statistical power of the analysis. Clearly in this case, statistical power improves markedly once the number of individual species-specific indicators reaches 30 or more.

Table 4. The number of individual species-specific indicators that need to meet their stipulated indicator-level targets in order to achieve the criterion-level target, implying that at the <5% probability significance level, a significant fraction of the individual indicator targets have been met. The table shows how variation in the number of species assessed affects the criterion level targets, and how modifying the level of ambition in the individual species targets affects the fraction of the assessed species that need to meet their indicator targets in order to demonstrate a significant improvement at the criterion level. Green shading shows the situation for the demersal fish analysed in the North Sea case study described in the working document.

<table>
<thead>
<tr>
<th>Number of individual species indicators</th>
<th>Conserving current status of sensitive ( k )-type species when at GES</th>
<th>Preventing undesirable change in ( r )-type opportunists at all times</th>
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<tbody>
<tr>
<td></td>
<td>Targets for individual indicators:</td>
<td>Targets for individual indicators:</td>
</tr>
<tr>
<td></td>
<td>&quot;The most recent indicator standardised deviate should equal or exceed the long-term time series mean (standardised deviate ≥0.0)&quot;</td>
<td>&quot;The most recent indicator standardised deviate should not exceed, and ideally be less than, the long-term time series mean (standardised deviate ≤0.0)&quot;</td>
</tr>
<tr>
<td>No active management</td>
<td>Active management</td>
<td>No active management</td>
</tr>
<tr>
<td>Number</td>
<td>Proportion</td>
<td>Number</td>
</tr>
<tr>
<td>--------</td>
<td>------------</td>
<td>--------</td>
</tr>
<tr>
<td>10</td>
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</tr>
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</tbody>
</table>

Regardless of whether the current management situation is "managing to achieve GES" or "managing to maintain GES", targets for opportunist \( r \)-type species should take account of the fact that increases in the abundance, biomass and distributional range of their populations are considered undesirable. Indicator-level and criterion-level targets similar to those suggested for conserving sensitive \( k \)-type species, albeit with the directionality reversed, would therefore seem appropriate. Thus individual indicator-level targets could be "the most recent indicator standardised deviate should not exceed, and ideally be less than, the long-term time series mean (standardised deviate ≤0.0)". The criterion level target would then be "the proportion of species
meeting their individual indicator targets should be sufficiently high that there is a <5% probability of this happening by chance". This binomial experiment is identical to the one described above, so the target numbers and proportions are the same (Table 4).

**Final caveats**

The two summation terms in equations 9 and 10 encompass the entire time series of each indicator in each new assessment, indicating that with each assessment the most recent indicator values added to the time series should be included when calculating the long-term mean and standard deviation. However it has also been suggested that both the mean and the standard deviation should be calculated up to a fixed point in time, eg 1983 to 2008, or perhaps up to the year that management measures are implemented.

Consider a situation where the fish community is below GES and management intent is to recover sensitive k-type species to achieve GES. For any given species the target is “the most recent indicator standardised deviate should exceed +0.5”. If in the most recent assessment, based for example on the period 1983 to 2008, the indicator standardised deviate just meets the target with a value of +0.500001. Then if in the following assessment, the summations to derive the mean and standard deviation are carried out over the same period (ie. 1983 to 2008), neither parameter will change, and if for the species in question there is no change in indicator value, it will again record an indicator standardised deviate of +0.500001 and be judged to have met its indicator level target. However, if currently below GES, the intent would be for year on year improvement until GES is ultimately reached. If in the initial assessment the criterion-level target is reached, so that significantly more than 35% of sensitive k-type species record indicator standardised deviates >=+0.5, this will inevitably cause their long-term time series means to have increased at the time of the next assessment. Under these circumstances, the species in the example above, showing no change in indicator value and the next, would in all likelihood record an indicator standardised deviate <+0.5, and consequently fail to meet its indicator-level target. Determining the means and standard deviations over the entire time series, including each new additional assessment, provides the “moving target” that ensures continual improvement towards the ultimate achievement of GES.

However the situation changes once GES is reached; now management intent would be to maintain current GES. For any given species the target is “the most recent indicator standardised deviate should equal or exceed the long-term time series mean (standardised deviate >=0.0)”. If in the most recent assessment, based for example on the period 1983 to 2008, the indicator standardised deviate fails the target with a value of -0.2. Then if in the following assessment, the summations to derive the mean and standard deviation are carried out over the same period (ie. 1983 to 2008), neither parameter will change, and if for the species in question there is no change in its indicator value, it will again record an indicator standardised deviate of -0.2, and will again be judged to have failed its indicator-level target. If nothing changes in each subsequent assessment, this species will be flagged as a perpetual “failure” case. However, if following the initial assessment, all subsequent assessments derive the means and standard deviations used to determine the indicator standardised deviates on the entire data set, then ultimately the mean values will fall and the species will get closer and closer to meeting its indicator-level target without any specific remedial action ever being implemented. This is not how the process should work.

The summation terms in equations 9 and 10 are therefore correctly stated if current status is “below GES” and management intent is to move towards GES. But if current status is “at GES” and management intent is to conserve current status, then $y_{last}$ should be replaced by $y_{GES}$, that year in which GES was deemed to have been attained, in the both equations.
As currently stated, even though the criterion-level targets might be met in successive assessments, particular sensitive $k$-type species might still be in decline. By chance alone, 35% of sensitive species could have indicator standardised deviates of $<-0.5$. With recovery management in place the proportion of “sensitive” species in this situation should be less than this. However, application of the binomial distribution, with 25 species and assuming a 65% probability of individual species specific indicator standardised deviates exceeding -0.5, even if criterion-level targets were being met at the 5% significance level, it is likely that 4 species (16%) will have indicator standardised deviates of $<-0.5$. If on each assessment occasion a different four species fall into this situation, then this would be of little concern. But if the same species perpetually meet this condition then this would infer that, although the status of sensitive $k$-type species was generally improving, particular species might still be in persistent decline, and so cause for concern. If the species involved should also happen to be included in, for example, the OSPAR list of threatened and endangered species (see Table 2), then this may need immediate action to be taken. The assessment process would need to be carried out in such a way as to flag-up these types of issue.

Similarly, in the process described thus far, no consideration has been given to the intermediate $r/k$ group of species. Whilst there is no underlying rationale to set targets for this group, nevertheless these species should still be assessed following the protocols described above so as to detect persistent changes in indicator values; particularly increasing trends in species nearer the $r$ end of the spectrum and decreasing trends in species nearer the $k$ end of the spectrum.
Summary of Indicator-level and Criterion-level Targets

**Distribution range**
The rationale for setting targets assumes that increasing human pressure would cause the ranges of opportunistic species to increase and of vulnerable species to decline. Table 5 lists both the individual indicator and the criterion targets proposed for the distributional range indicators defined for two different marine situations; continental shelf seas and shelf-edge seas.
Table 5. Indicators and targets proposed for the "distribution range" indicator applied to demersal fish.

<table>
<thead>
<tr>
<th>Situation</th>
<th>Indicator metric</th>
<th>Species type</th>
<th>Management situation</th>
<th>Indicator targets</th>
<th>Criterion target</th>
</tr>
</thead>
<tbody>
<tr>
<td>Continental shelf sea</td>
<td>Proportion of sampled ICES rectangles in which the species occurs.</td>
<td>Opportunistic r-type species</td>
<td>At any time</td>
<td>The most recent standardised deviate of the distribution range indicator should not exceed, and ideally be less than, the long-term time series mean (standardised deviate ≤0.0).</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Sensitive k-type species</td>
<td>Below GES</td>
<td>The most recent standardised deviate of the distribution range indicator should exceed +0.5.</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>At GES</td>
<td>The most recent standardised deviate of the distribution range indicator should equal or exceed the long-term time series mean (standardised deviate ≥0.0).</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
<tr>
<td>Shelf-edge sea</td>
<td>Proportion of depth bands in which the species occurs.</td>
<td>Opportunistic r-type species</td>
<td>At any time</td>
<td>The most recent standardised deviate of the distribution range indicator should not exceed, and ideally be less than, the long-term time series mean (standardised deviate ≤0.0).</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Sensitive k-type species</td>
<td>Below GES</td>
<td>The most recent standardised deviate of the distribution range indicator should exceed +0.5.</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>At GES</td>
<td>The most recent standardised deviate of the distribution range indicator should equal or exceed the long-term time series mean (standardised deviate ≥0.0).</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
</tbody>
</table>
Distributional pattern within the range
Assuming that some sort of dispersal process (e.g. the Ideal Free Distribution [Fretwell and Lucas, 1970; Partridge, 1978]) underlies most fish distributions, the following rationale for target setting was initially considered. The core preferred habitat of any organism is a physical attribute of the environment, and therefore remains constant in size. Being the best habitat for the species in question, this is where abundances are highest. Like any predator, fisheries are attracted to such locations because high abundances are reflected in higher catch rates per unit effort. Thus fish are primarily extracted from such locations, reducing abundance in these prime habitat hot-spots, tending to reduce contagion and causing the mean:variance ratio to increase. As abundance declines in the prime habitats, the Ideal Free model predicts that individuals in less optimal habitats will move into the now vacant, or less occupied, prime locations. This will tend to reduce the extent of the species range as the lowest occupancy, least optimal, locations empty; the number of high density sites (ICES rectangles) remains the same, but the number of low density sites reduces. The effect of this on the mean is much less than on the variance, consequently causing the mean:variance ratio to rise, again suggesting increased dispersion across the populated range.

However, the arguments presented above are primarily based on how we might expect the degree of contagion/dispersion to vary mainly as a function of changes in abundance and range. On this premise, the level of correlation between the distributional pattern indicator and indicators of abundance and distribution range would be high; implying a high level of indicator redundancy and negating the need for all three indicators. It was not clear how distributional pattern might change independently of these other two indicators. Furthermore, indicators of abundance, biomass and distribution range were thought likely to be more sensitive to anthropogenic pressure than the proposed indicator of distributional pattern. Finally, mitigation measures implemented to meet other targets, particularly those set for other ecosystem components, such as habitat restoration and revovery of prey resources, might lead to conflicts between targets set for fish distributional pattern. Consequently, no targets are currently proposed for the distributional pattern indicator. This indicator needs further development in respect of the fish component.

Population abundance and/or biomass
In setting targets, we have assumed that the pressure from human activity would cause a decrease in the abundance and biomass of vulnerable species and an increase in the abundance and biomass of opportunistic species. Management would therefore need to reverse or control these tendencies. Table 6 lists both the individual indicator-level and the criterion-level targets proposed for both population abundance and population biomass indicators.
Table 6. Indicators and targets proposed for the population abundance and biomass indicators applied to demersal fish.

<table>
<thead>
<tr>
<th>Situation</th>
<th>Indicator metric</th>
<th>Species type</th>
<th>Management situation</th>
<th>Indicator targets</th>
<th>Criterion target</th>
</tr>
</thead>
<tbody>
<tr>
<td>Continental shelf and shelf-edge seas</td>
<td>Population abundance</td>
<td>Opportunistic r-type species</td>
<td>At any time</td>
<td>The most recent standardised deviate of the population abundance and population biomass indicators should not exceed, and ideally be less than, the long-term time series mean (standardised deviate ≤0.0).</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
<tr>
<td></td>
<td>AND Population biomass</td>
<td>Sensitive k-type species</td>
<td>Below GES</td>
<td>The most recent standardised deviate of the population abundance and population biomass indicators should exceed +0.5.</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
<tr>
<td></td>
<td>density</td>
<td></td>
<td>At GES</td>
<td>The most recent standardised deviate of the population abundance and population biomass indicators should equal or exceed the long-term time series mean (standardised deviate ≥0.0).</td>
<td>The proportion of species meeting their individual indicator targets should be sufficiently high that, based on the binomial distribution, there is a &lt;5% probability of this happening by chance.</td>
</tr>
</tbody>
</table>

**Population demographic characteristics**

It was not obvious what the targets should be for such an indicator. Initially one might expect that, for a vulnerable species, any reduction in mortality rate should increase life-expectancy so that a greater proportion of each population should grow to exceed their length at first maturity, and so cause indicator values to increase. However, reduced fishing activity should also reduce discarding rates of juvenile fish. Increased survival rates among such fish might bring about an initial decline in the proportion of mature fish indicator, and it is only when these fish eventually grow to exceed each species’ length at first maturity threshold that indicator values might start to increase. Some relatively simple population modelling is required to explore these questions before appropriate targets can be set. The results of such modelling should also indicate whether it is better to derive this indicator based on biomass or on abundance. Consequently, no targets are currently proposed for the population demographic characteristics indicator. This indicator needs further development in respect of the fish component.
Composition and relative proportions of ecosystem components

Long lags between changes in pressure and structural and compositional responses by fish communities have been demonstrated empirically in several studies (Daan et al., 2005; Greenstreet et al., 2011; Shephard et al., In press) and seem to be supported by the results of process-based, multiple-species, size-based modelling work (Fung et al., in review; Rossberg et al., 2008). The North Sea case study analysis has indicated that the increase in species richness might be interpreted as either a positive or a negative response to changes in fishing pressure depending on the length of the response lag involved (Figure 2). Alternatively the change in species richness may be the result of change in environmental conditions.

Fishing generally targets the more dominant species in the community. Reductions in dominant species’ population size might be expected to bring about an increase in species evenness; declining status could be associated with increased evenness! Conversely, reduced predation pressure from target species could elicit marked expansion in the populations of their smaller prey species, particularly among the competitively dominant species. Trophodynamics theory suggests that increases in prey biomass should be approaching an order of magnitude larger than declines in predator biomass; in terms of abundance, this difference would be even greater. Thus across the whole community, this marked increase in the apparent dominance of a few prey species would bring about a reduction in species evenness associated with fishing disturbance; the process alluded to in several previous studies (Greenstreet and Hall, 1996; Greenstreet et al., 1999; Greenstreet and Rogers, 2006; Greenstreet et al., in press). But the above hypothesis posits opposing responses in species evenness to variation in fishing pressure, suggesting that these metrics should be applied independently to different size classes of fish in the community, with specific targets set for each size class dependent on their trophic functional role.

Once again, more multi-species community modelling work is required to provide a secure basis for setting targets for these indicators. In recent years there has been a proliferation in the number of such models available, and their development has become increasingly directed towards addressing such management-related questions. Within the next year it is quite possible that some of these models could be sufficiently developed as to be able to support the target setting process. It is almost certain that they will be available by 2015 to provide advice in respect of the measures required to achieve such targets.

The LFI was developed specifically to assess the impact of fishing on the “health” of the demersal fish community. As such the LFI is perhaps the best metric currently available to fulfil the indicator role for this Criterion. The use of the LFI to support the OSPAR EcoQQ for fish communities has been fully developed (Greenstreet et al., 2011; Shephard et al., In press) and we simply adopt the indicator definition and targets. Since a single indicator for each “fish community” assessed is proposed, the indicator-level target is de facto the same as the criterion-level target (Table 7).
Table 7. Indicators and targets proposed for the composition and relative proportions criterion in respect of demersal fish.

<table>
<thead>
<tr>
<th>Situation</th>
<th>Indicator metric</th>
<th>Species type</th>
<th>Management situation</th>
<th>Indicator targets</th>
<th>Criterion target</th>
</tr>
</thead>
<tbody>
<tr>
<td>Continental shelf and shelf-edge seas</td>
<td>The “large fish indicator” (LFI)</td>
<td>Demersal assemblage (but with specified species composition)</td>
<td>At any time</td>
<td>LFI &gt; regional/survey specified target.</td>
<td>Indicator target should be met</td>
</tr>
<tr>
<td></td>
<td>“the proportion of fish (by weight) exceeding a specified length threshold”</td>
<td></td>
<td></td>
<td>North Sea target 0.3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Length thresholds are dependent on region/survey/species suite</td>
<td></td>
<td></td>
<td>Celtic Sea target 0.4</td>
<td></td>
</tr>
<tr>
<td></td>
<td>North Sea threshold 40cm</td>
<td></td>
<td></td>
<td>Targets are dependent on region/survey/species suite</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Celtic Sea threshold 50cm</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Large fish (by weight)**

In the North Sea there is some evidence to suggest that the link between variation in the LFI and changes in the trophic structure of the food web might not be as tight as traditional size-based aquatic food web theory might predict. In a heavily fished region the expected trend towards smaller size was noted, but no change in trophic level (Nitrogen stable isotope ratio analysis) was observed (Jennings et al., 2002); large piscivores of one species were replaced by small piscivores of another species. The use of the LFI as a food web structure (proportion of top predators) indicator therefore still needs to be properly validated. However, The EC 2010 Decision document explicitly stipulates that the LFI should be used as a Descriptor 4 food web indicator to address Criterion 4.2. *Proportion of selected species at the top of food webs.* Until this validation work is done, this is the proposed course of action. The indicators and targets are identical to those proposed for Criterion 1.7 *Composition and relative proportions of ecosystem components* given in Table 7.

**Abundance trends of functionally important selected groups/species**

Although trophic functional groups for fish have been suggested, the actual assignment of species and size classes to these groups has yet to be carried out. So while these indicators are conceptually easy to derive, no operation indicators are currently available. Neither is it clear what the appropriate targets should be. The fish communities of the North Sea and Celtic Sea have undergone quite profound changes in composition and structure that these have been described as regime shifts (Greenstreet et al., 1997; Heath, 2005a; 2005b). The causes of these changes have been linked to both anthropogenic and climatic factors. Considerable development is therefore required before this indicator can be made operational and appropriate targets selected.

**Making the species-specific targets operational**

Targets suggested for the species-specific indicators considered operational at the current time are all essentially “trend-based”; the most recent indicator value should be at a particular level with respect to all other indicator values in the time series. The problem with such targets is that they do not actually define GES in respect of each particular indicator. Rather they describe how the indicator should change under different circumstances and management regimes; whether currently at GES and managing to conserve this state, or currently below GES and managing to attain this state. If currently in the latter position, there is no mechanism associated with each individual indicator to determine when or if GES is actually reached, and the situation should
switch to the former. Choice of target in Tables 17 and 18 is dependent on the condition given in the “Management situation” column. However, nothing linked to the indicators themselves provides any indication as to which condition currently prevails.

The LFI is the only indicator proposed so far for the fish component that has an explicit numeric target value. Implicit in this is the assumption that target values for the LFI represent GES; if the target for any given region is met then GES for fish in that region will have been met in respect of this indicator. At the last assessments, the LFI in the North Sea was 0.22, below the target of 0.3 (Greenstreet et al., 2011) and the LFI for the Celtic Sea was 0.12, below the target of 0.4 (Shephard et al., in press). According to the LFI therefore, GES for the fish community in both regions has yet to be achieved. Consequently, a precautionary approach would be to assume that, with regard to the species-specific indicators listed in tables 17 and 18, the current management situation is “below GES”, and the corresponding indicator-level and criterion-level targets should be adopted. If in subsequent assessments these targets are achieved, this would not mean that the state of the fish community was necessarily at GES. It would, however, imply that the management measures introduced to achieve GES were being effective, and were actually altering the state of the community in the desired direction towards GES.

The corollary to this proposition is that, once the LFI targets are met, GES is achieved. Targets for the species-specific indicators associated with the management situation of “at GES”, listed in Tables 17 and 18, should then be adopted. This essentially identifies the LFI as the indicator that defines GES for the broader fish community, but at present there appears little alternative to this. Interestingly, this means that when LFI targets are reached, values of the species-specific indicators that prevail at that time might now reflect actual GES values. There is some circularity to this argument, but again at present it seems that this cannot be avoided.

The accompanying North Case study working document that has informed much of the development of indicators and targets for fish just considers demersal fish species. Pelagic species (herring, sprat, sandeels, etc) are a major component of fish communities so this could be considered to represent a major gap. However, groundfish surveys also sample pelagic species and fisheries management certainly uses these data to derive abundance indices for species such as herring and sprat. The shoaling nature of herring and other pelagic species affects sampling probabilities in a way that differs markedly from the sampling probability of most demersal species. Two survey trawls in a given ICES rectangle may miss all shoals of herring in the area, so recording a rectangle density estimate of zero. Conversely, both trawls might hit herring shoals, thereby recording a density far in excess of the actual density present in the rectangle. Demersal species do not shoal to the same extent; they are more evenly distributed so trawl sample densities are generally more representative of actual local scale density and this issue is less of a concern. This variability in sampling probability can markedly influence outcomes in studies applying univariate community metrics to groundfish survey data. Hence, in such studies, pelagic species have traditionally been excluded from the analysis (Greenstreet et al., 1999; Greenstreet & Rogers, 2006; Greenstreet et al., 2011). However it is a question of variability and scale. In surveys covering whole marine regions, such as the ICES Q1 IBTS covering most of the North Sea, this sampling variability evens out; hit the shoals in one rectangle, miss them in the next. At a regional scale therefore, the overall estimate of density may reasonably well reflect the actual density of fish across the whole region, but the inter-rectangle variability will be high. Thus groundfish surveys may well be capable of providing data for pelagic species to support the species-specific indicators proposed here. However, while these indicators may be reliable enough at large spatial scales, MSFD sub-region scale (eg.the Greater North Sea), they may be less reliable at smaller spatial scales.
There is some concern over the issue of indicator redundancy (Greenstreet et al., in press). For example among the fish indicators for which targets are being proposed, there is a strong possibility that distribution range, population abundance and biomass will all be strongly cross-correlated. Should one indicator be selected, perhaps the one deemed most sensitive to changes in pressure, or the one thought most comprehensible to a layman audience, or the one where the evidence base underpinning target setting is most comprehensive? Or should a concept of “headline” indicators and supporting “tracking” or “surveillance” indicators be invoked? Or should all of them just be proposed anyway?

Finally there is the issue of indicator compatibility. If in recovering populations of sensitive $k$-type species this brings about an increase in large bodied piscivorous fish in the community, what effect might this have on management capacity to achieve indicator targets proposed for other ecosystem components? For example, maintenance of kittiwake chick productivity might be compromised through increased competition between adult kittiwakes, foraging for fish prey for their chicks, and increased numbers of piscivorous fish predators in the marine environment.

References


Fung, T., Farnsworth, K. D., Shephard, S., Reid, D. G., and Rossberg, A. G. In review. Recovery of community size-structure from fishing requires multiple decades. ICES Journal of Marine Science,


ANNEX 7 Detailed targets and indicators for each biodiversity descriptor
ANNEX 8 - CBA spreadsheets of biodiversity targets, pressures and measures:

Biodiv_Descriptors_in
fo_CBA_FINAL_subsequent amendments_in Yellow_incl_sediments_v3.xls

Addendum:

This spreadsheet was submitted to Eftec (working under contract to Cefas) on 03/06/11 as part of the MSFD CBA. Since then, there have been some changes to the targets of the biodiversity descriptors and the other Descriptors referred to in the spreadsheet, however, the essence of the targets remains the same.

With regard to costings for additional monitoring of pelagic habitat – these have been amended but not updated in the spreadsheet – please refer to section 4.11 of this report.

Notes on spreadsheets

1. Criterion Targets (Column E)
These are derived from the amalgamated outputs of the HBDSEG Birmingham workshop (29th-31st March) and further development at the Kew workshop (4th/5th May 2011). Each of these targets represents the boundary between achieving GES and failing to meet GES. Column F refers to the level of confidence that the proposed criterion target would equate to GES. Where the confidence in the criterion target equating to GES is low (i.e. where there is low confidence in the indicator targets underpinning these, and/or if the underlying data are sparse), the target may need to be set higher in terms of required state in order to ensure that GES is reached. Determining whether or not these criterion level targets have been reached (using the aggregated data on the underlying indicator targets) will ultimately determine whether GES has been achieved for UK biodiversity under the descriptors 1, 4 and 6.

2. Costs of monitoring progress towards criterion targets
The monitoring costs are also derived from the amalgamated outputs of the HBDSEG Birmingham and Kew workshops. They represent a summary of the estimated cost of monitoring the indicators underpinning each criterion. We have indicated where existing monitoring will suffice (i.e. No additional cost), and where additional monitoring or new monitoring schemes are required at medium cost (<£100k pa) and a high cost (>£100k pa). These cost categories were originally suggested by the MMO, but HBDSEG has included more precise cost estimates where possible.

3. Pressure and Impacts
For each biodiversity target, we have identified (in Column J) which MSFD pressures and impacts will most greatly affect the probability of attaining them (from Table 2 Annex III of the Directive - see worksheet <MSFD Pressures & Impacts>). We have included only those pressures that are
listed as having a high or medium impacts on the component - see worksheet *<Pressures on components>* which was derived from outputs from the Cefas workshop on Measures.

HBDSEG were invited by eftec to make additions to this list of pressures, where necessary. HBDSEG have assigned a confidence rating for the link between the listed pressures and the changes in state which would effect achieving the specified criterion target (Column K).

4. Pressure Targets
For each pressure/impact identified, we have assigned a relevant pressure target(s) from the Pressure Descriptors (3, 5, 8, 9, 10 and 11) (see current provisional list in worksheet *<Cefas Pressure targets>*); where none of these were appropriate, we have included new pressure targets suggested by the HBDSEG workshop. Where no pressure targets are appropriate or available from either source, columns L & M were left blank.

5. Measures
For each target, we have inserted a list of likely management measures that would help achieve that criteria targets (Column N). The measures list comes from the Cefas pressures workshop, and is presented in the worksheet *<Cefas Measures>*. At Eftec’s recommendation, HBDSEG have included additional measures where there were felt to be gaps. In column O, we have specified the confidence associated with the effectiveness of the measure(s) to achieve the target. The confidence rating relates to the ‘suite’ of measures proposed for each pressure, although HBDSEG has provided further information where confidence ratings vary significantly within each ‘suite’.

6. Business as Usual (BAU)
In column P, we have assessed whether the criterion targets will be attained by 2020 under a business as usual scenario of measures or whether additional measures and response monitoring will be required. The BAU scenario was informed by Chapter 2 ‘Baseline’ in the ‘ME5405: Cost-Benefit Analysis of MSFD Targets – Draft Interim Report’. Again, the measures in column N were considered as a ‘suite’. Column Q refers to additional measures and monitoring required over and above BAU.

7. Definitions of levels of confidence and uncertainty

The definition of ‘confidence’ is based on Intergovernmental Panel on Climate Change (2005). A *level of confidence is used in the IA to describe uncertainty that is based on expert judgment (in terms of the correctness of an analysis or a statement). Definitions of the terms used to communicate this are provided in Table A.1.*

<table>
<thead>
<tr>
<th>Terminology</th>
<th>Degree of confidence in being correct</th>
</tr>
</thead>
<tbody>
<tr>
<td>Very High confidence</td>
<td>At least 9 out of 10 chance of being correct</td>
</tr>
<tr>
<td>High confidence</td>
<td>About 8 out of 10 chance</td>
</tr>
<tr>
<td>Medium confidence</td>
<td>About 5 out of 10 chance</td>
</tr>
<tr>
<td>Low confidence</td>
<td>About 2 out of 10 chance</td>
</tr>
<tr>
<td>Very low confidence</td>
<td>Less than 1 out of 10 chance</td>
</tr>
</tbody>
</table>
8. Key unknowns
The information provided to eftec in this spreadsheet is affected by a series of ‘unknowns’. Specifically, there are uncertainties surrounding:

The UK MPA network:
- Percentage cover of final UK MPA network
- Management measures in final MPA network
- Likely displacement of activities when MPAs are designated - What will be required over and above the MPA network to protect species and habitats

EU Directives (HD, BD, WFD)
- The alignment of MSFD targets with those in other existing EC Directives

Quality vs quantity
- GES is unlikely to be met uniformly across UK seas. It is anticipated that the status of some areas may get worse, while others better. GES targets will therefore need to take into consideration issues of scale, specifically regarding acceptable proportions of areas under different condition and their contribution to GES.

Current Status
- For some features/areas, we have little information regarding current state in relation to the target state.

Overlaps
- In most cases, targets will be achieved through the implementation of multiple measures. It is challenging to distinguish between the relative importance of each measure at this stage. There are also significant overlaps/redundancies in relation to monitoring costs.

Accuracy
- Amalgamation in terms of both within a criterion (i.e. combining indicator targets to come up with criterion targets) and across biodiversity components (targets applying to multiple biodiversity components) may significantly affect the accuracy of cost estimations.