

# ICES WGECO REPORT 2016

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## Report of the Working Group on the Ecosystem Effects of Fishing Activities (WGECO)

6–13 April 2016

Copenhagen, Denmark



**ICES**  
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International Council for  
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## Contents

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Executive Summary .....	4
<b>1 Opening of the meeting.....</b>	<b>6</b>
<b>2 ToRs for the 2016 meeting.....</b>	<b>7</b>
<b>3 Develop and assess indicators of Good Environmental Status for seabed integrity: Further develop indices of impact on the seabed and sensitivity of the benthic community to different pressures, and the link to function (ToR a) .....</b>	<b>8</b>
3.1 Exposure: Overlap Habitat and Fishing pressure .....	9
3.1.1 Canadian case study .....	9
3.1.2 North Sea case study .....	16
3.2 Effect of Fishing: Habitat sensitivity .....	16
3.2.1 Canadian case study .....	16
3.2.2 Sensitivity matrix .....	17
3.2.3 OSPAR Benthic Habitat Indicator (BH3) .....	18
3.2.4 Approach based on mortality and recovery .....	20
3.2.5 Approach based on longevity .....	24
3.2.6 Possible improvements in assessing habitat sensitivity .....	25
3.3 Discussion .....	27
3.3.1 Relevance in the context of WKFBI .....	28
3.4 References .....	30
<b>4 Investigate possible indicators of scavengers, examine their relation to discard amounts and evaluate the spatial effect of a landing obligation on the scavengers (ToR b) .....</b>	<b>34</b>
4.1 Southern North Sea .....	36
4.2 Northern North Sea .....	39
4.3 Conclusions and plans for progress .....	40
4.4 References .....	41
<b>5 Use the data available to evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced” (ToR c).....</b>	<b>43</b>
5.1 Examine time-trends in stocks that are rebuilding and dependent stocks .....	43
5.1.1 Methods.....	43
5.1.2 Results .....	44
5.1.3 Conclusions.....	48
5.1.4 Future work .....	48
5.2 Establish the distribution of total catch among species (catch species dominance curves).....	48

5.2.1	Methods.....	48
5.2.2	Results .....	50
5.2.3	Conclusions.....	54
5.2.4	Future work.....	54
5.3	References .....	55
<b>6</b>	<b>Distributional indicators in the context of MSFD: developing operational and surveillance indicators and exploring the link to pressure and drivers (tor d) and request from ICES to determine appropriate methods used to identify significant distribution shifts (ToR h) .....</b>	<b>56</b>
6.1	Key aspects of distribution which are relevant in the context of biodiversity and associated metrics .....	56
6.1.1	Caveats related to sampling coverage .....	58
6.1.2	Estimating distributional indices from data.....	58
6.2	Surveillance indicators for the MSFD .....	63
6.3	ICES Special Request.....	64
6.4	References .....	66
<b>7</b>	<b>Review and further develop indicators of state of sensitive species throughout the ICES area and suggest risk based advice on sensitive species (ToR e).....</b>	<b>71</b>
7.1	Estimating missing life-history parameter values.....	73
7.2	Deriving the Greenstreet et al. (2012) ‘sensitivity’ metric .....	84
7.3	A ‘sensitivity’ metric that takes parameter interactions into account .....	86
7.4	Evaluating the data paucity issue.....	88
7.5	References .....	89
<b>8</b>	<b>Consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level (in collaboration with the DEVOTES project and building on work from WGBIOV) (ToR f) and the request ‘In the light of previous advice, review the guidance on the most suitable and defensible approach to aggregate species within species groups (such as birds, mammals, reptiles, fish and cephalopods), for the state assessments of the MSFD that was developed by the Workshop on providing a method to aggregate species within species groups for the assessment of GES for MSFD D1 (WKD1Agg)’ .....</b>	<b>92</b>
8.1	Review the guidance developed by the Workshop on providing a method to aggregate species within species groups for the assessment of GES for MSFD D1 (WKD1Agg) .....	92
8.2	Consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level; comparing WKD1Agg, WKGESFish and WGEKO approaches.....	95
8.3	References .....	99
<b>Annex 1:</b>	<b>List of participants.....</b>	<b>100</b>

<b>Annex 2:</b>	<b>Agenda .....</b>	<b>102</b>
<b>Annex 3:</b>	<b>WGECO terms of reference for the next meeting .....</b>	<b>105</b>
<b>Annex 4:</b>	<b>Recommendations to other EGs .....</b>	<b>108</b>

## Executive Summary

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The 2016 meeting of WGECO was held at the ICES HQ in Copenhagen, Denmark from 6–13 April 2015. The meeting was attended by 17 delegates from 13 countries, and was chaired by Anna Rindorf (Denmark). The work conducted was centred on six Terms of Reference and two advisory requests concerning indicators of Good Environmental Status (GES) of the benthic community, the potential effect of a landing obligation on the benthic ecosystem, the degree to which fisheries are balanced and possible effects of rebuilding predatory stocks, indicators of distribution change, the definition of sensitive species, and the integration of indicators into GES at the descriptor or ecosystem component level.

WGECO continued the work to **develop and assess indicators of Good Environmental Status of the benthic community**. The concept of risk (= Exposure\*Effect) was used to assess the risk of impact based on fishing pressure and habitat (or benthic) sensitivity. Examples for each of these two aspects, i.e. Exposure (represented by the overlap between the pressure and habitat maps) and Effect (determined by some measure of habitat sensitivity in relation to the pressure) were presented. In order to fit the framework of the forthcoming workshop on benthic indicators (WKFBI), “good practices” and “lessons learned” relevant to the WKFBI ToRs were identified. Canadian and North Sea case studies were used as examples where two approaches were assessed for determining sensitivity *a posteriori* and other approaches reviewed.

WGECO continued **evaluating the effect of a landing obligation on scavengers of the benthic ecosystem**. Distribution data were compiled for discarded biomass from the Discardless project, epibenthic scavenger abundances and swept-area estimates from the BENTHIS project and the probability of occurrence of hagfish from FishBase. The analysis did not show a demonstrable link between discarded biomass and either scavengers as a proportion of the benthic community or individual “scavenger” species abundance. The analysis was conducted at ICES Rectangle scale, and it was concluded that it may be more successful at a finer scale of resolution than at the level of the ICES rectangle and with the inclusion of carrion produced by fisheries in the trawl path. These suggestions will be addressed in next year’s work.

Six predator fish species that have sustained rebuilding of biomass over periods of at least eight years were identified in the work **using the data available to evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced”**. Of 20 dependent prey species only one, shrimp, exhibited a decline that could be attributed to its predator, Iceland cod. WGECO concluded that there is little empirical evidence that rebuilding of piscivorous fish stocks has led to declines in dependent prey species. WGECO has advanced methodology to empirically estimate the degree of balance of fisheries in relation to available biomass. Based on this methodology, the fisheries in Iceland and the Bay of Biscay are more balanced than those in the North Sea and New England.

Key aspects of **species distribution** in relation to pressure and climatic drivers were identified and metrics for each aspect identified. These metrics cannot replace a detailed investigation of the spatial distribution but can be used to scan a large number of species distribution for trends and indications of relationships with pressure and climate metrics. Metrics of distribution in the context of the Marine Strategy Framework Directive (MSFD) were discussed.

A complex modelling approach to **defining suites of 'sensitive' fish species** was developed to derive more accurate parameter estimates where these data were absent, and a second 'sensitivity' metric was designed that explicitly used the relationships between the four life-history parameters. Because of the large amount of missing data that needed to be modelled to support both 'sensitivity' metrics, the potential of simply using each species' maximum recorded length as an alternative metric to define 'sensitivity' was explored. Such data are available for all species, thereby eliminating the need for the complex estimation modelling. However, it is likely that 15% to 20% of the species deemed to be 'sensitive' by the two metrics, would be classified as 'resilient' on the basis of maximum recorded length alone.

WGECO reviewed the report of the Workshop on providing a method to aggregate species within species groups for the assessment of GES for MSFD D1 (WKD1Agg). This group listed advantages and drawbacks of integration at different levels, and listed some available integration methods. Choice of approach ultimately pertains to policy decision rather than science. Proposed integration approaches are also appropriate to other MSFD descriptors such as D3, D4 and D6.

## 1 Opening of the meeting

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The **Working Group on the Ecosystem Effects of Fishing Activities (WGECO)** met at ICES, Denmark from 6–13 April 2016. The list of participants and contact details are given in Annex 1. The chair, Anna Rindorf (Denmark) welcomed the participants and highlighted the variety of ToRs. The draft agenda was presented (Annex 2) and Terms of Reference for the meeting (see Section 2) were discussed. One additional ToR was received during the meeting. A plan of action was adopted with individuals providing presentations on particular issues and allocated separate tasks to begin work on all ToRs.



## 2 ToRs for the 2016 meeting

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The **Working Group on the Ecosystem Effects of Fishing Activities** (WGEKO), chaired by Anna Rindorf, Denmark, met in Copenhagen, Denmark 6–13 April 2016 to:

- a) Develop and assess indicators of Good Environmental Status for seabed integrity: Further develop indices of impact on the seabed and sensitivity of the benthic community to different pressures, and the link to function (joint ToR with BEWG);
- b) Investigate possible indicators of scavengers, examine their relation to discard amounts and evaluate the spatial effect of a landing obligation on the scavengers;
- c) Use the data available to evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced”;
- d) Distributional indicators in the context of MSFD: developing operational and surveillance indicators and exploring the link to pressure and drivers (*this ToR may be supplemented by a request to point to the most appropriate indicators to detect distributional change*);
- e) Review and further develop indicators of state of sensitive species throughout the ICES area and suggest risk based advice on sensitive species;
- f) Consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level (in collaboration with the DEVOTES project and building on work from WGBIOV);
- g) In the light of previous advice, review the guidance on the most suitable and defensible approach to aggregate species within species groups (such as birds, mammals, reptiles, fish and cephalopods), for the state assessments of the MSFD that was developed by the Workshop on providing a method to aggregate species within species groups for the assessment of GES for MSFD D1 (WKD1Agg);
- h) Determine appropriate methods to identify significant distribution shifts.

WGEKO reported by 24 April 2016 to the attention of the Advisory Committee.

### **3 Develop and assess indicators of Good Environmental Status for seabed integrity: Further develop indices of impact on the seabed and sensitivity of the benthic community to different pressures, and the link to function (ToR a)**

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This ToR was interpreted and developed by WGECO such that it can inform the forthcoming ICES Workshop on Fisheries Benthic Impact (WKFBI) and notably their ToRs:

- b) Evaluate information on sensitivity of the benthic community of the various seabed habitats that ensures habitat maps for sensitivity can be produced for at least one demonstration area of NW European waters (MSFD region/subregion).
- c) Evaluate impact maps that combine the benthic information on sensitivity and fishing pressure maps (fishing abrasion, weight and value of landed catch), taking into account differences in benthic impact of the various fishing gears / métiers.
- d) Using the workshop, evaluate and synthesis findings (ToR a–c) aimed at tangible use of indicators of the state of the seabed in relation to fishing pressure.
- e) Prepare a guidance document on how pressure maps of fishing intensity contribute to an assessment of the state of seabed habitats. This should include “principles and good practices” that can be used when regionally operationalizing indicators to assess the impact of fishing to the seabed.

The guidance provided for the WKFBI distinguishes specific steps that are required for assessing how pressure maps of fishing intensity contribute to an assessment of the state of seabed habitats:

- 1) Acquire a habitat map covering as much of the MSFD region as possible. The thematic classes of the habitat map need to be aligned or cross-referenced to the classes used in the sensitivity assessment without significant gaps.
- 2) Acquire sensitivity information for each thematic class of habitat to surface/subsurface abrasion. Habitat map polygons are then attributed with the sensitivity code.
- 3) Acquire surface/subsurface abrasion layers (pressures from fishing activity) for the MSFD assessment area. Clip layers to match the habitat map coverage.
- 4) Combine (intersect/raster calculator/map algebra) the attributed habitat map with the abrasion layers. Use a combination matrix (categorical attribution of pressure and sensitivity) to combine sensitivity and pressure to calculate impact.
- 5) Map the impact of fishing on benthic habitat.
- 6) Extract summary statistics/indicators from the impact map. Produce a confidence assessment for the map and summary statistics.

Where relevant we refer to each step in our text to facilitate cross-referencing with the WKFBI agenda.

The concept of risk (=Exposure\*Effect) is useful for the quantification of the (risk of) impact based on fishing pressure and habitat (or benthic) sensitivity (Piet *et al.*, in prep.). Exposure is the likelihood of occurrence and magnitude of fishing pressure and the effect of fishing is determined by the sensitivity of the habitat and the type and intensity of fishing (Stelzenmüller *et al.*, 2010). Sensitivity is defined as the degree to which this ecosystem component responds to a particular pressure, taking into consideration the mortality of this component induced by this pressure and the time it would take to recover (resilience) (Tillin *et al.*, 2006; Tyler-Walters *et al.*, 2001; Hope, 2006; Halpern *et al.*, 2008). Thus the state of the seabed, i.e. seabed integrity, is determined by (1) the exposure of the seabed to fishing pressure and (2) an understanding of the effect fishing has on the state of the seabed. Fishing impact is the difference in seabed integrity as the result of fishing.

Below we provide worked examples for each of these two aspects, i.e. Exposure (represented by the overlap between the pressure and habitat maps) and Effect (determined by some measure of habitat sensitivity in relation to the pressure). Based on the information available to WGECO on these two aspects we identify “good practices” and “lessons learned” that are relevant to the WKFBI process. We appreciate that this process is locked into the use of physical habitat maps as proxies for biological communities and provide a case study that used a similar methodological approach as that proposed by WKFBI but assessed biological habitats directly. These approaches are contrasted in the discussion Section 3.3.

### 3.1 Exposure: Overlap Habitat and Fishing pressure

This aspect of risk fits mostly with ToR c of WKFBI requiring to combine existing habitat maps with maps of fishing pressure. However, we note that the habitat maps to be considered are largely constructs of surficial geology and depth. While both of these are important determinants of species distributions, ultimately the sensitivity of a habitat type will depend on the life-history traits of the composite species or species groups. Consequently, assessments of impact (steps 4–6) can only be conjectural. However, they may serve to locate areas of concern that require a second suite of area-specific analyses.

Here we present two approaches that were applied to determine the exposure of the seabed to fishing:

- Canadian case study
- North Sea case study

#### 3.1.1 Canadian case study

Fisheries and Oceans, Canada (DFO) has undertaken an evaluation of Significant Benthic Areas (SBAs) on its east coast from the US border in the south to the Eastern Arctic. SBAs are defined in DFO’s Ecological Risk Assessment Framework (ERAF) as “significant areas of cold-water corals and sponge dominated communities” (<http://www.dfo-mpo.gc.ca/fm-gp/peches-fisheries/fish-ren-peche/sff-cpd/risk-ecolo-risque-eng.htm>). The taxa analysed were sponges (Porifera), large and small gorgonian corals (Alcyonacea), and sea pens (Pennatulacea) which are also considered vulnerable marine ecosystem indicators in the adjacent international waters falling in the Northwest Atlantic Fisheries Organization (NAFO) Regulatory Area. NAFO re-

viewed over 500 benthic invertebrate taxa against the Food and Agricultural Organization of the United Nations (FAO) criteria (FAO, 2009) identify the VME indicators which in addition to the above included non-coral and sponge taxa.

Lesson Learned: Combine species with similar morphologies and habitat requirements.

Hui *et al.* (2013) compared the predictive performance of species groups against separate species distribution models (SDMs) using a number of multispecies datasets. They found that using groups of species with similar environmental requirements improved model accuracy and discriminatory capacity compared to separate SDMs. Further, the approach was specifically endorsed for rare species. Dunstan *et al.* (2013) outlines an approach for grouping species with similar environmental requirements based on regression models.

### 3.1.1.1 Identification of Significant Benthic Areas (Steps 1 and 2)

Maps of the location of significant concentrations of corals and sponges were produced through quantitative analyses of research vessel trawl survey data, supplemented with other data sources where available. Those analyses were conducted following a bio-regionalization approach in order to facilitate modelling of similar species, given that many of the multispecies surveys do not record coral and sponge catch at species level resolution.

Good Practices: Conduct modelling within biogeographic units.

Identifying regions of comparative homogeneity in species composition and performing models within those areas is very important when working with data identified only to high level taxonomic units (e.g. Porifera in this case study). Foster *et al.* (2013) outline a statistical approach for achieving this using presence-absence data from multiple species and co-located environmental data. Bioregions were identified by areas where the vector of probabilities of observing a set of species remains approximately constant within a region and distinct between regions. In the Canadian case study, expert opinion was used to generate the biogeographic zones applied (DFO, 2009).

There are a number of problems associated with using trawl biomass data of non-target benthic taxa including: biomass of corals and sponges are often associated with low and variable catchability to survey trawl gear; taxonomic resolution of the catch is often variable and poor; the biomass distribution may be shaped by previous fishing activity; number and biomass of highly aggregated species typically bear no relationship with trawl length and so standardization of catches is not straightforward; the precision of the sampling location is not known (often >1 km). Presence-absence models were thought to reduce catchability issues associated with trawl-caught biomass data.

Good Practices: Conduct separate analyses of biomass data from different survey fishing gears.

Maps based on biomass data should be constructed separately for different gear types unless the frequency distributions can be shown statistically to be identical (Kenchington *et al.*, 2014).

In the Canadian case study, a combination of models was used each with a different theoretical basis. Kernel density analysis utilizes spatially explicit data to model the

distribution of a variable of interest. It is a simple non-parametric neighbour-based smoothing function that relies on few assumptions about the structure of the observed data. It has been used in ecology to identify hot spots, that is, areas of relatively high biomass/abundance. It does not extrapolate to areas where there is no data, it does not use null data, and at most it interpolates to the extent of the search radius (in their examples <25 km). The trawl survey data were used by first applying kernel density estimation (KDE) to create a modelled biomass surface for each taxon, and then applying an aerial expansion method to identify thresholds for significant concentrations from the broader distribution (Kenchington *et al.*, 2014).

Good Practices: Use biological, ecological, statistical or environmental based thresholds for delineating areas to reduce subjectivity and assess the biological implications of decisions.

In the case study, thresholds based on change in area were used to identify aggregations of habitat-forming taxa.

Species Distribution Models (SDMs) were then applied to complement the KDE analyses. Even simple environmental variables such as depth and slope can give greater ecological insight into the models and create more realistic maps. Random Forest (RF) (Breiman, 2001) is a non-parametric machine learning technique, where multiple regression or classification trees (usually  $\geq 500$ ) are built using random subsets of the data. For classification with presence-absence response data, random forest can be used to predict the probability of a species' presence in non-sampled areas by identifying areas with similar environmental conditions. RF is a robust statistical method requiring no distributional assumptions on covariate relation to the response compared with other classical statistical models such as generalized linear models (GLM) or generalized additive models (GAM). It can handle a large amount of input variables effectively without variable deletion and can also account for correlation as well as interactions among variables. For each bioregion data from over 100 environmental variables were obtained from a broad range of data sources and spatially interpolated using geostatistical methods. For each variable the underlying data distribution and relevant diagnostics of the interpolation models were presented and the predicted surfaces with their associated error surface displayed (e.g. Beazley *et al.*, 2016). Only variables that had good diagnostics were used in the SDM analyses, leaving 54 to 76 environmental variables as predictors. In the case of regression using biomass response data, random forest can predict the species' biomass distribution. Biomass (kg) data associated with the DFO trawl survey records were also extracted and used in random forest modelling. In order to avoid the introduction of bias related to differences in catchability between gear types, biomass regression random forest models were run separately on each of the three gear types when applicable.

Lessons Learned: Use appropriate models and compare results using different modelling approaches to evaluate the robustness of the maps.

Many different SDM models and modelling approaches are currently available (Guisan and Zimmerman, 2000). There are over 60 methods to choose from, including geostatistical interpolators (e.g. kriging), non-geostatistical interpolators (e.g. inverse distance weighting, natural neighbours, nearest neighbours), and methods that combine both (Li and Heap, 2008). In recent years, machine learning methods have become increasingly common. These comprise of a series of non-parametric techniques capable of synthesizing regression or classification functions based on the available data. Random Forest (RF) (Breiman, 2001) is one such method that can be used for regression or

classifications and is considered superior to most commonly used methods (Cutler *et al.*, 2007). Others have used regression based techniques such as generalized additive (GAM) modelling (e.g. Rooper *et al.*, 2014) which offer a different theoretical basis for the production of the interpolated surfaces.

**Good Practices:** Identify sources of error in all data and data layers and assess their spatial pattern. When possible, provide error maps and quantify uncertainty.

Continuous interpolated surfaces produced at high resolution often show very detailed spatial variation, implying that the surfaces are very precise. However, spatial interpolation methods are affected by sample size, sampling design and data quality properties, and variation within the data has very large impacts on the performance of the spatial interpolators (Li and Heap, 2008). Identify sources of uncertainty and undertake methods to minimize it or include it in the modelling process to give realistic measures of confidence around predictions (Beale and Lennon, 2012).

**Lessons Learned:** Consider model boundaries to be fuzzy.

While RF models are more robust against overfitting compared to other machine learning algorithms such as bagging, they have been observed to overfit when data contains very "noisy" classification or regression tasks (Segal, 2004). In the SDM context, random forests make distinct spatial predictions compared to GAMs and GLMs. In this regard, overfitted RF models can make predicted distribution maps very "patchy" at smaller spatial scale and difficult to interpret (Franklin, 2010). Caution should be made to not over-analyse predicted distribution maps at smaller spatial scales.

The Canadian case study found that classification random forest models generated using all presence and absence data (i.e. unbalanced species prevalence) and a threshold equal to species prevalence produced the most realistic presence probability prediction surfaces and highest model accuracy in instances when the input data were highly imbalanced and spatially biased across the study area. Random down-sampling of the absence data often resulted in gross extrapolation of high presence probability beyond the location of presence observations. This was likely exacerbated when down-sampling to match a small number of presence observations. However, nearly identical presence probability surfaces and model accuracy measures between balanced and unbalanced runs were produced when there were a high and relatively even number of presence and absence observations across the study extent.

A hierarchical modelling approach was followed in the Canadian case study. Biomass data were used to perform the KDE analyses. Then presence-absence SDMs were performed using RF to extrapolate to unsampled areas within the environmental data range of the response variables. RF regression trees were run to produce predicted biomass surfaces to compare with the KDE-derived significant concentration areas. The KDE-derived polygons were overlain on the presence-absence SDM (Figure 3.1) to allow for modifications of the former. Lastly, in some instances, the top predictor variables in the regression RF trees were used in generalized additive models (GAMs), to evaluate biomass surfaces using a different theoretical basis (machine learning vs. regression; Figure 3.2). Models were evaluated whenever possible using independent data sources such as local ecological knowledge records, the NOAA Deep-Sea Coral Data Portal, those from other scientific research and surveys, and commercial fisheries observer data. Validation was performed for each taxonomic group in each region.

Good Practices: Validate models whenever possible using independent data.

Model validation should be undertaken and any geo-referenced data type can be overlain on the model surfaces, including commercial fishing data, underwater camera surveys, research surveys, traditional and/or local ecological knowledge (TEK/LEK), etc.

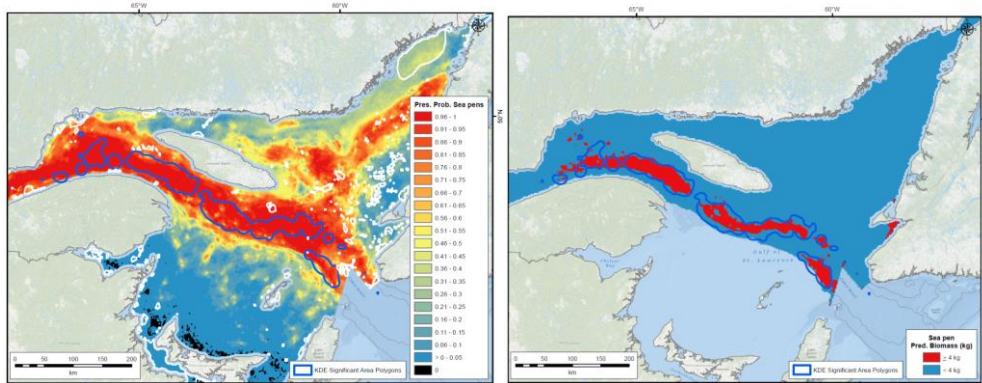


Figure 3.1. Left: Overlay of sea pen KDE-derived polygons (blue) on the random forest presence-absence prediction map (1 km<sup>2</sup> resolution) for the Gulf of St Lawrence. The polygons were generally consistent with the prevalence map. Note that areas of extrapolation are outlined in white. Right: Predictions of biomass ( $\geq 4$  kg) of seapens above (red) and below (blue) the threshold (4 kg per tow) of significant concentrations of seapens identified by the KDE analysis (from Murillo *et al.*, 2016).

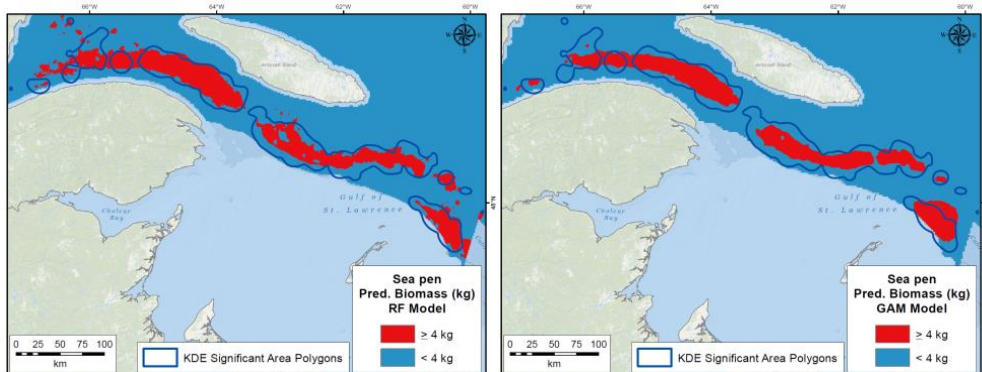


Figure 3.2. Predictions of biomass ( $\geq 4$  kg) of seapens based on RF models (left panel) and GAM models (right panel) above and below the threshold of significant concentrations of sea pens identified by KDE ( $\geq 4$  kg per tow) in the northern portion of the Gulf of St Lawrence. Areas of significant concentrations of sea pens identified by KDE are shown in blue outline, while predicted biomass  $\geq 4$  kg is shown as solid red areas. In this region the northern and southern areas were analysed separately for the biomass models as the two areas were sampled using different trawl gears (from Murillo *et al.*, 2016).

The biomass models outlined in this case study did not consider the effect of disturbance by human activities on the data as noted above. Predicted distribution and biomass can therefore be confounded by fishing activities, and areas that are physically suitable (from presence-absence models) but are predicted to have low occurrence or biomass may not necessarily indicate bad model performance. The taxa considered in this report are vulnerable marine ecosystem (VME) indicators and are highly aggregating, structure-forming megafaunal groups that can be found in ‘significant concentrations’ constituting VMEs (Kenchington *et al.*, 2014). The life-history traits of

these species, such as slow growth rates, late age of maturity, or their structural complexity make them very vulnerable to fishing activities (FAO, 2009). In order to consider how anthropogenic pressure has influenced these ecosystems, a measure of this, such as fishing intensity, should be included as a predictor variable in the RF models and the effects of changes in the pressure explored (Bergstrom *et al.*, 2013). This kind of analysis would point out potential species distribution and could indicate areas for future restoration initiatives. Foster *et al.* (2014) demonstrate how the cumulative effects of fishing on fish assemblages in Australia change as the fishery developed. They applied a model-based clustering method (Dunstan *et al.*, 2013; Hui *et al.*, 2013) to identify species with common environmental responses (species archetypes) and produced mostly interpretable ecologically sensible responses by the archetypes to the fishing pressure.

### 3.1.1.2 Distribution and Intensity of Fishing Effort (Step 3)

In the Canadian Case Study the distribution and intensity of fishing effort during the period 2005–2014 was estimated on the basis of two data sources: logbook information and Vessel Monitoring System (VMS) data (Koen-Alonso *et al.*, 2016). Logbooks are filled out by fishers during fishing trips and contain details on the vessel, effort and catch characteristics. VMS data are positional information that are transmitted automatically at regular intervals via satellite from fishing vessels. Data from VMS provide high resolution positions recorded at higher frequencies compared to logbook reporting, however, not all fisheries use VMS and its use varies by region, gear type, directed species, and vessel size.

Because gear impacts from different fisheries vary greatly, effort was grouped into categories with similar gears and fishing behaviours. A total of 13 fisheries classes were defined, which encompass 98% of all the fishing effort recorded in fisheries logbooks across Atlantic Canada and the Eastern Arctic. Some of these classes correspond to well-defined fisheries (e.g. shrimp), while others represent aggregates that encapsulate some general features across several fisheries (e.g. pelagic).

Given the diversity of gears and modes of operation, the unit of effort considered for logbook data in this study was vessel-day (VD) where one fishing location is displayed for a given vessel-day of fishing, which allows for general comparisons across fisheries classes. Where data were available, the footprint of fisheries classes was estimated by plotting fishing locations on a 1 km x 1 km grid, and the intensity of the fishing effort was calculated by the cumulated number of fishing observations within each cell of the grid. This allows defining the areal extent of the fishing operations, the footprint, and the intensity of use of different regions within that footprint.

For comparative purposes, effort was standardized by converting fishing intensities to percentiles using vessel-days for logbooks, and hours fished per unit area for VMS. Percentiles were calculated by summing the total effort a given fishery exerted in each 1 km x 1 km cell, ranking cells by descending effort, and calculating the cumulative percentage of the total effort exerted by the entire fishery. The 20th percentile of fishing effort represents the area where the top 20% heaviest fishing effort took place, the 40th percentile represents the area where the top 40% heaviest fishing took place, and up to the 100th percentile where 100% percent of the effort takes place. This procedure allows identifying hot spots of fishing activities and, by making effort relative to the total within each data source (logbooks and VMS), also provides a natural way for integration of these data sources.

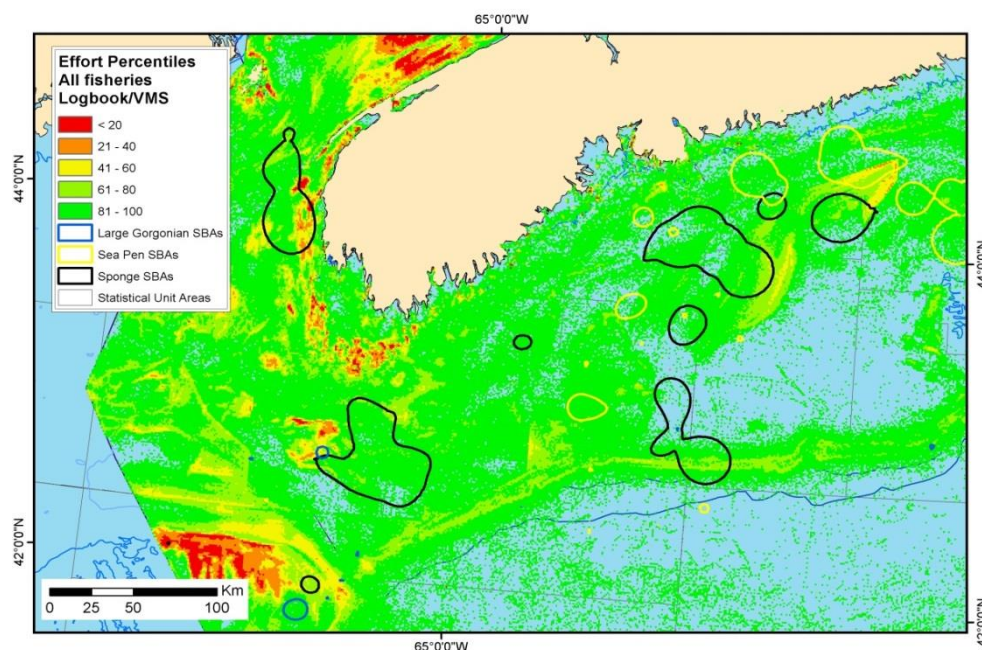


### 3.1.1.3 Overlap between Significant Benthic Areas and Fishing Activities (Steps 4, 5, 6)

Within each bioregion, overlap between Significant Benthic Areas and fishing activities was evaluated in two ways:

- 1) the percentage of the fishing footprint that overlaps Significant Benthic Areas; and
- 2) the percentage of Significant Benthic Area that is fished.

These analyses were made for logbook and VMS data independently as well as using a combined logbook / VMS effort layer (Figure 3.3). In those cases where VMS coverage is high, footprint and overlap using VMS data is considered a better estimate for these indices than those calculated from logbooks. The results of these analyses were used to identify areas of potential conservation and management concern.



**Figure 3.3.** Preliminary map of the overlap between fishing effort from all fisheries and significant areas of sponges (KDE-derived black polygons) and seapens (KDE-derived yellow polygons) off southwest Nova Scotia, Canada (M. Koen-Alonso, pers. comm.).

**Lessons Learned:** Avoid fishery-specific analyses whenever possible; instead try to integrate all fisheries classes operating in a given bioregion within a single assessment.

Depending on the nature of the fishing activities (e.g. target species, gear used, mode of operation), it is possible that the overlap between SBAs and fishing activities is dominated by a single fisheries class, but in most cases, the total overlap with fishing activities will be the result of the cumulative action of multiple fisheries classes operating in the same area. Performing independent overlap analyses for each individual fisheries class may lead to underestimate the actual overlap between sensitive benthic areas (SBAs) and fishing, and could make more difficult/complex the process of designing and implementing conservation measures for the protection of SBAs.

**Good Practices:** Integrate all fisheries classes within a single overlap assessment, classifying the fisheries into classes which are expected to reflect rea-

sonably consistent impact types on the seabed (as opposed to fisheries classes that simply reflect fisheries regulations).

This approach allows the overlap assessment to become a suitable platform for subsequent impact analyses, which typically characterize risk as a function of the overlap, frequency of operation, and impact type (e.g. associating impact levels to the gear used and modes of operation).

### 3.1.2 North Sea case study

The North Sea is one of the few European marine regions for which all the information required for such an approach is available, i.e. a basin-wide coverage of a seabed habitat map (based on EUSeaMap) distinguishing the main (EUNIS level 3) habitats and annual fishing intensity maps (period 2010–2012) at appropriate spatial scale, covering and distinguishing, all the main fleet segments that disturb the seabed. For the spatial scale they applied grid cells of 1 x 1 minute (approximately 2 km<sup>2</sup>), the temporal scale was annual but other spatio-temporal scales may be applied.

Fishing pressure affecting the seabed (i.e. Physical Damage) is usually expressed as the extent of surface area that is threatened by those segments of the fleet that use towed fishing gears and the literature distinguishes several potential indicators that capture this (Piet and Hintzen, 2012; Fock *et al.*, 2011; Rijnsdorp *et al.*, in press). It is also well established that bottom trawling is patchy and that this patchiness needs to be taken into account to assess the impact of trawling on the benthic ecosystem (Rijnsdorp *et al.*, 1998; Lee *et al.*, 2010a; Ellis *et al.*, 2014). Therefore all of these indicators involve the calculation of a surface area that is considered significantly impacted based on estimates of fishing intensity per grid cell, for example: the surface area disturbed by fishing (i.e. summation of all grid cells in which trawling was recorded), surface area disturbed more than a specific threshold (e.g. frequency >1 yr<sup>-1</sup>) or the summed surface area of the most intensively trawled grid cells (e.g. encompassing 90% of the annual fishing effort). Calculation of fishing intensity was based on the swept-area, i.e. footprint, as calculated by Eigaard *et al.* (2015) for all major European fishing gears.

## 3.2 Effect of Fishing: Habitat sensitivity

If a habitat and its benthic community are exposed to fishing, then the effect of fishing on the state of the seabed will be determined by the sensitivity of the habitat and its benthic community to the fishing activity. Here we provide several examples of how sensitivity can be estimated and applied to determine how fishing affects the state of the seabed:

- Canadian case study
- Sensitivity matrix
- Benthic Habitat Indicator 3
- Approach based on mortality and recovery
- Approach based on longevity

### 3.2.1 Canadian case study

Species that are extremely sensitive to fishing disturbance will be vulnerable to very light intensity of fishing with high levels of destruction on the first pass (e.g. Figure 3.4). For such species the degree (intensity, duration) of disturbance may only be rel-

evant with respect to the spatial footprint of the contact zone with the sensitive species. Conversely, less sensitive species would show varying responses to both intensity and duration but always in a neutral or negative trajectory. Recovery would also be determined by life-history traits, with fragile but rapid growing and/or fecund species, such as some sponges, recovering faster than fragile but slow growing species with low fecundity and variable recruitment such as some corals. Equally, less fragile species could be split into similar traits groups and show different recovery trajectories. Consequently, application of trait-based groupings for rapid and slower impact and recovery can disentangle community responses and clarify non-intuitive outcomes (e.g. light fishing has high impact and vice versa).



**Figure 3.4.** Dead gorgonian corals (*Prinnoa resedaeformis*) caught up in fishing gear (live colonies shown in inset). Photo is from the Northeast Channel off southwest Nova Scotia, Canada.

Foster *et al.* (2014) provide an elegant example of how species traits of fish species respond to cumulative fishing pressure. Using groups of species with similar responses to environmental variables (archetypes) they demonstrated that the archetype showing greatest decline in abundance was made up of species that had the highest mean values of generation time, oldest age at maturity and longest lifespan. However, life-history traits were only indicative of response and there was sufficient variability such that responses to fishing pressure were not always readily predicted.

### 3.2.2 Sensitivity matrix

This approach involves a sensitivity assessment, based on expert judgement, which evaluates the physical damage and mortality associated with any bottom fishing activity that interacts with seabed habitats. A sensitivity matrix is developed that summarizes all potential pressure attributes on species and/or habitats and the magnitude of the impact is scored qualitatively. An example of a sensitivity matrix is shown in Table 3.1. Numerous sensitivity assessment approaches have been developed, but they are generally quite similar (Roberts *et al.*, 2010). Eno *et al.* (2013) evaluated the sensitivity of the combination of resistance and recovery of 31 habitat types to damage by 14 categories of fishing activity. Tyler-Walters *et al.* (2009) adopted the Marine Life Information Network (MARLIN), <http://www.marlin.ac.uk/> to evaluate the sensitivity of sedimentary environments to fishing. The MB0102 approach evaluates the sensitivity for EUNIS Level 3 broad-scale habitats, OSPAR threatened and/or declining

habitats and species to pressures (Tillin *et al.*, 2010). The “Features Activity Sensitivity Tool” (FeAST), uses the MB0102 approach to assess sensitivity of species and habitats to pressures in Scottish waters (<http://www.marine.scotland.gov.uk/FEAST/>). The “Marine Evidence based Sensitivity Assessment” (MarESA) framework evolved from the MARLIN and the MB0102 approaches (Tillin *et al.*, 2010; Tillin and Tyler-Walters, 2014). The objective of this framework was to reduce the uncertainty associated with the sensitivity assessment of selected subtidal sedimentary habitats by assessing sensitivity at a finer scale compared to the MB0102 approach. This involved sensitivity assessment for selected UK Level 5 biotopes and some Level 6 biotopes, including circalittoral biotopes, seagrass habitats, *Mytilus edulis* beds and *Sabellaria spinulosa* reefs (Tillin and Tyler-Walters, 2014). The sensitivity was also evaluated for “ecological groups”, which were defined based on the Level 5 and 6 biotopes (Tillin and Tyler-Walters, 2014), with each group consisting of ecologically similar species. This approach enabled reduction in the number of sensitivity assessments while retaining the different elements of the biological assemblage (Tillin and Tyler-Walters, 2014). In 2016, WGDEC evaluated the use of MarESA sensitivity methods and reviewed the Level 4 deep-sea habitats that were considered to need sensitivity assessment. Their conclusion was to score all deep-sea habitats as being of “high” sensitivity, while they acknowledged that some habitats were more sensitive than others (WGDEC, 2016).

### 3.2.3 OSPAR Benthic Habitat Indicator (BH3)

OSPAR is developing the Benthic Habitat Indicator 3 (BH3) indicator, aimed at computing an index of the spatial extent of impacts of a particular pressure. In 2015, OSPAR requested ICES to produce fishing abrasion pressure maps within the OSPAR maritime area in order to support the development of the BH3 indicator ([http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2015/Special\\_Requests/OSPAR\\_biodiversity\\_indicators\\_benthic\\_habitats\\_1.6.6.3.pdf](http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2015/Special_Requests/OSPAR_biodiversity_indicators_benthic_habitats_1.6.6.3.pdf)). Within ICES, the Benthic Ecology Working Group (BEWG, 2015), the Working Group on Marine Habitat Mapping (WGMHM, 2015), the Working Group on Spatial Fisheries Data (WGSFD, 2015) and the workshop on guidance for the review of descriptor six (WKGMSFDD6, 2015) have all reviewed or been actively working on this ICES request. WGSFD (2015) has produced maps that show surface abrasion (damage to seabed surface features) and subsurface abrasion (penetration and/or disturbance of the substratum below the surface of the seabed) using preprocessed VMS fishing data to calculate swept-area by fishing gears. Such maps have been produced for the Greater North Sea, Celtic Seas and the Bay of Biscay/Iberian Peninsula regions, and there are plans to include other maritime areas in the North Atlantic (WGSFD, 2015). In 2016, the biodiversity committee developed a four step analytical framework for the BH3 indicator (OSPAR, 2016). The first step involved evaluation of the spatial distribution of the pressure. In step two and three the sensitivity of habitats to pressures is evaluated spatially for the OSPAR maritime area. The last step involved evaluation of the cumulative physical disturbance to a particular habitat (OSPAR, 2016).

**Table 3.1. Example of sensitivity scoring matrix.**

PRESSURE THEME			
Pressure Broadscale Habitats	Penetration and/or disturbance of the substratum below the surface of the seabed	Shallow abrasion/penetration: damage to seabed surface and penetration	Surface abrasion: damage to seabed surface features
Pressure Benchmarks	Structural damage to seabed >25 mm	Damage to seabed surface and penetration ≤25 mm	Damage to seabed surface features
Deep-sea bed	H (L)	H (L)	H (L)
Deep-sea mud	H (M)	H (M)	NS-H (M-H)
Infralittoral rock	NA (L)	NA (L)	NA (L)
Circalittoral rock	NA (L)	NA (L)	NA (L)
Subtidal coarse sediment	M	L	NS
Subtidal coarse sediment (<50m)	H	M	M
Subtidal gravel beds	L	M	H
Subtidal macrophyte-dominated sediment	H	H	H
Subtidal mixed sediments	M	M	L
Subtidal mixed sediments (<50m)	M	M	M (L)
Subtidal mud	M (L)	M (L)	L-M (L)
Subtidal mud (<50m)	H	M	M
Subtidal sand	M	M	L
Subtidal sand <50m)	H	M	L
Vents, seeps, hypoxic and anoxic habitats of the deep sea	NA (L)	NA (L)	NA (L)
Subtidal macrophyte-dominated sediment	H	H	H
Subtidal biogenic reefs	M	H	H
Deep-sea bioherms	H (H)	H (H)	H (H)

### 3.2.4 Approach based on mortality and recovery

This approach is used in the BENTHIS project to calculate seabed integrity and how this impacted by fishing is based on the notion that habitat sensitivity is determined by two parameters: Instantaneous mortality by the passing of the gear and recovery (Piet *et al.*, in prep). This approach is based on the concept of risk involving exposure and effect. This method is applied in the North Sea. For the spatial scale we applied grid cells of 1x1 minute (approximately 2 km<sup>2</sup>), the temporal scale was annual but other spatio-temporal scales may be applied.

The method follows the DPSIR analytical framework often applied for EBM (Atkins *et al.*, 2011; Knights *et al.*, 2013) and specifically EBFM (Martins *et al.*, 2012) but applies an ERA to estimate Pressure, State, Impact and propose a management Response. The risk that seabed integrity is compromised is based on our knowledge of the exposure of the seabed to fishing pressure and an understanding of the effect fishing has on the state of the seabed. The management response is based on a fisheries credit system that applies SIQ to mitigate fishing impact on the seabed.

#### 3.2.4.1 Habitat sensitivity

Following (Jennings *et al.*, 2012) this study defines habitat sensitivity as the time (RT) that it takes for the benthic community biomass to recover to a specific level of the unimpacted biomass ( $x\%B_0$ ) following a single pass of the fishing gear (see Figure 3.5). The effect of a single pass of the fishing gear on the habitat is captured by two parameters, i.e. the Instantaneous Mortality IM (here only caused by physical damage) and the Recovery Time (RT), which are habitat- and gear-specific (Figure 3.5). Values for IM and RT were taken from Pitcher *et al.* (in prep.), based on data presented in Collie *et al.* (2000) (see Table 3.2).

Table 3.2. Estimated Instantaneous Mortality (*IM*) and Recovery Time (*RT*) values for the different gear habitat combinations and the sustainable trawling frequencies that is calculated based on these parameters and on what is considered a “significant deterioration of quality” (SDQ) determining the level of unimpacted biomass to which the benthic community should recover. The *IM* and *RT* estimates are from Piet *et al.* (in prep.) and the SDQ estimates from Pitcher *et al.*, in preparation).

Habitat	Gear	IM	RT	SUSTAINABLE TRAWLING FREQUENCY YR <sup>-1</sup> DEPENDING ON “SIGNIFICANT DETERIORATION OF QUALITY” (SDQ)			
				80%	90%	95%	99%
Biogenic	OT	0.39	3.03	0.48	0.18	0.08	0.02
Gravel	OT	0.48	3.03	0.41	0.14	0.06	0.01
Sand	OT	0.37	15.59	2.68	1.02	0.42	0.06
Mud	OT	0.27	6.39	1.44	0.51	0.19	0.03
Biogenic	BT	0.45	3.03	0.4	0.16	0.07	0.01
Gravel	BT	0.53	3.03	0.35	0.14	0.06	0.01
Sand	BT	0.43	15.59	2.15	0.91	0.43	0.08
Mud	BT	0.33	6.39	1.2	0.49	0.21	0.04
Biogenic	TD	0.67	3.03	0.26	0.1	0.04	0.01
Gravel	TD	0.72	3.03	0.25	0.1	0.05	0.01
Sand	TD	0.66	15.59	1.67	0.66	0.24	0.04
Mud	TD	0.61	6.39	0.71	0.28	0.12	0.02

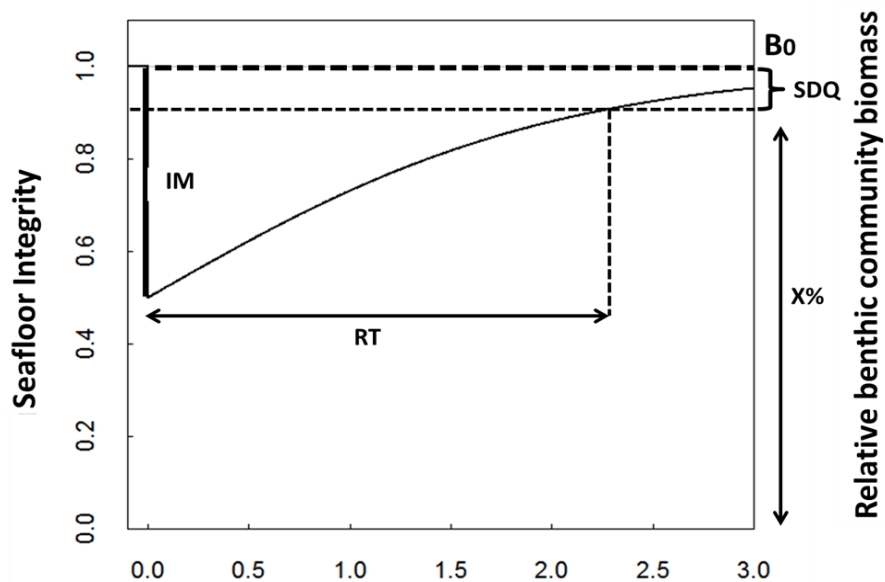


Figure 3.5. Schematic of the parameters: Impact Mortality (*IM*) caused by trawling, recovery time (*RT*), and the “significant deterioration of quality” (SDQ) which determines the level of the unimpacted biomass ( $x\%B_0$ , see Table 3.2) that determines the *RT*.



### 3.2.4.2 State expressed as Seabed integrity

For the main EU policy framework, i.e. the Marine Strategy Framework Directive (MSFD) the state of the seabed is captured by the phrase seabed integrity. Seabed integrity ( $SI_{g,t}$ ), at some point in space (grid cell  $g$ ) and time ( $t$ ) is determined by the habitat sensitivity and (historic) fishing intensity  $FI_{g,t-1}$ . The state of the seabed  $SI_{g,t}$ , can be estimated as an equilibrium biomass value (Figure 3.5), expressed as a fraction of unimpacted biomass ( $B_0$ = carrying capacity) using the following equation:

$$SI_{g,t+1} = SI_{g,t} * \left[ 1 - \left( \frac{HFI_{g,t+1} * IM_g}{RT_g} \right) \right]$$

where

$$SI_{g,t+1}=0 \text{ if } HFI_{g,t+1} > RT_g/IM_g,$$

Where  $SI_{g,t}$  is the seabed integrity at the onset of the year  $t$ . Hence  $SI_{g,0}=1$  as the seabed was unimpacted with biomass at  $B_0$ .  $HFI_{g,t}$  is the historic fishing intensity over the (longer term) period prior to the year  $t+1$  for which  $SI_{g,t+1}$  will be calculated. The length of this period on which the calculation of  $HFI$  is based should be chosen such that it adequately represents the fishing disturbance over a time-scale appropriate to that habitat. In practice this may be limited by data availability. For this study it was limited to only one year in order to allow the consistent calculation of SIQ (see below) in two of the three years for which data covering the entire North Sea were available. Thus, although in this study the  $HFI$  period is effectively only one year we chose to name it historic fishing intensity to emphasize longer periods should probably be used if longer time-series become available. The parameters  $RT$  and the  $IM$  (see Table 3.2) depend on the gear-habitat combination in each grid cell  $g$ .

The above equation predicts a linear relationship between HFI and SI for an individual species. Observations of the relationship between community biomass and fishing, however, generally show an exponential decline (Hiddink *et al.*, 2006), and this discrepancy may be due to the variation in recovery potential and carrying capacity that are found within natural communities. Some species grow fast, some grow slow, and some can achieve high population biomasses while others only achieve low biomasses. Here we assume that the  $RT$  and  $B_0$  values have an exponential distribution, with the sum of  $B_0$  within the community summing to 1, and the mean of  $RT$  being equal to the  $RT$  in Table 3.2. Values of  $B_0$  and  $RT$  were randomly chosen for 1000 species, and the effect of fishing on the SI of each species was modelled using the above equation. The community SI was simply the sum of the SI's of all individual species. The relationship between SI and HFI for this community was similar to an exponential decline.

The status of the seabed in each grid cell can then be aggregated into a seabed integrity metric  $SI_t$  for the entire MSFD (sub)region  $G$  according to:

$$\overline{SI}_t = \frac{\sum_{g \in [1,G]} SI_{g,t}}{G}$$

Figure 3.6 shows how a specific long-term pressure (i.e. type of gear fished at a specific trawling frequency) will result in a specific decrease in quality (dependent on the sensitivity of the habitat). Table 3.2 shows how allowing recovery to x% of unimpacted biomass  $B_0$ , e.g. 80% $B_0$ , 90% $B_0$  or 95% $B_0$  determines which trawling frequencies can be considered compatible with GES. In the most common habitat in the North Sea, i.e. Sublittoral sand covering almost 60% of the area, a 95% threshold would allow a



patch to be fished with a beam trawl less than once every two years (Sustainable trawling frequency  $<0.43 \text{ yr}^{-1}$ ). In contrast, in case of the application of an otter trawl (OT) in a gravel habitat, this same 95% deterioration in quality threshold would determine any fishing intensity  $<0.06 \text{ yr}^{-1}$  compatible with GES.

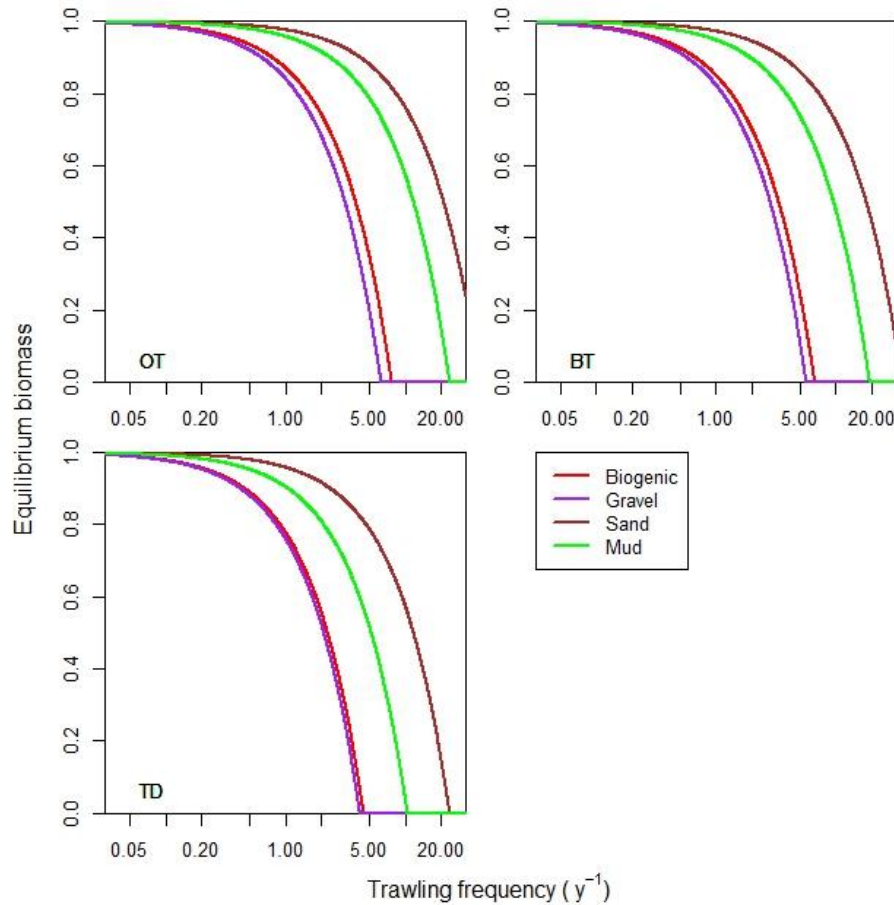


Figure 3.6. A risk-based measure of seabed integrity reflecting Seabed Integrity (=Equilibrium total biomass of the benthic community relative to the unimpacted biomass  $B_0$ ) at different trawling intensities for three trawling gears (OT=Otter trawl, BT=Beam trawl and TD=Dredge) and four habitats.

### 3.2.4.3 Fishing pressure affecting the seabed

Fishing pressure affecting the seabed (i.e. Physical Damage) is usually expressed as the extent of surface area that is significantly impacted by specific segments of the fleet that use towed gears (Piet and Hintzen, 2012; Fock *et al.*, 2011; Rijnsdorp *et al.*, in press). It is well established that bottom trawling is patchy and that this patchiness needs to be taken into account to assess the impact of trawling on the benthic ecosystem (Rijnsdorp *et al.*, 1998; Lee *et al.*, 2010a; Ellis *et al.*, 2014). Therefore all of these are based on estimates of fishing intensity per grid cell, for example: the surface area disturbed by fishing (i.e. summation of all grid cells in which trawling was recorded), surface area disturbed more than a specific threshold (e.g. frequency  $>1 \text{ yr}^{-1}$ ) or the summed surface area of the most intensively trawled grid cells (e.g. encompassing 90% of the annual fishing effort). Our calculation of fishing intensity was based on the swept-area, i.e. footprint, as calculated by Eigaard *et al.* (2015) for all major European fishing gears.

#### 3.2.4.4 Fishing impact on the seabed

In this study the seabed integrity in one year minus the impact due to Physical Damage (PDI) gives the seabed integrity in the next year. Thus

$$PDI_{g,t} = SI_{g,t} - SI_{g,t+1}$$

This could result in  $PDI_{g,t} < 0$  which would imply that seabed integrity is improving because the recovery rate of the benthic community is larger than the fishing-induced physical damage rate. Similar to  $SI$  the  $PDI_{g,t}$ 's per grid cell can be aggregated into a summary statistic  $PDI_t$  for the entire MSFD (sub)region. A positive  $PDI_t$  indicates that seabed integrity is deteriorating.

Fishing pressure and impact can also be calculated for a specified subset of the fishing activity, e.g. a particular fleet segment, fishing vessel or trip, in order to guide EBM aimed at reducing the fishing pressure and/or impact on the seabed. This partial fishing pressure then reflects the relative contribution of this subset, i.e. fleet segment, fishing vessel or trip, to the overall pressure or impact.

#### 3.2.5 Approach based on longevity

In this approach from the BENTHIS project, the sensitivity of the seabed is estimated from the longevity distribution of the benthic community that is typical for a seabed habitat (Figure 3.7). The impact of bottom trawling on seabed was estimated by combining trawling intensity with the longevity distribution of the benthic community. If the reciprocal of the trawling intensity, which reflects the average time interval between two successive trawling events, is less than the lifespan of an organism, the integrity of the seabed habitat to provide a place to live for the organism will be compromised. If the biomass of the benthic taxa shows a log-linear relationship with the longevity of its taxa, the seabed integrity in a grid cell can be estimated as the biomass proportion of the benthic community.

where the reciprocal of the trawling intensity ( $\frac{1}{\tau}$ ) is larger than the longevity of the taxa:

$$SBI = \exp(\alpha + \beta \left(\ln \frac{1}{\tau}\right)) / (1 + \exp(\alpha + \beta \left(\ln \frac{1}{\tau}\right)))$$

$\alpha$  and  $\beta$  are the coefficients of the logistic regression of the cumulative biomass against the  $\log_e$  of the lifespan of the taxa.

The seabed integrity of a habitat or management area can be obtained by adding up the seabed integrity indices over the grid cells and dividing by the surface area of the habitat or management area.

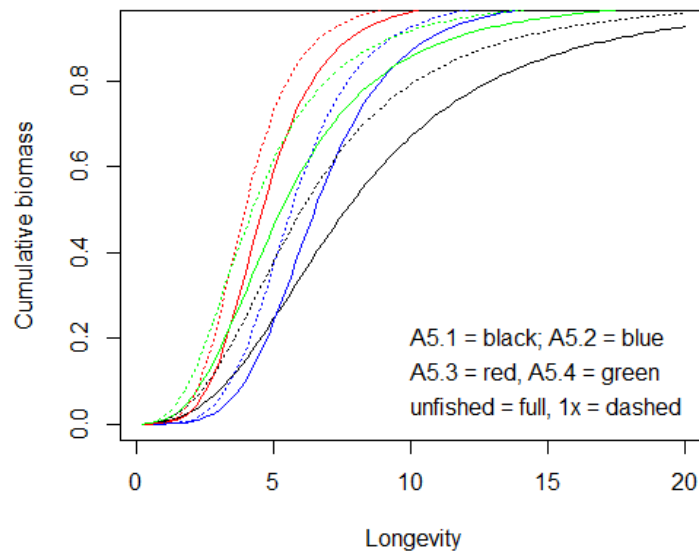


Figure 3.7. Cumulative biomass in relation to the longevity of the taxa for four EUNIS-3 habitats, sublittoral coarse sediment (A5.1): sublittoral sand (A5.2) sublittoral mud (A5.3) and sublittoral mixed sediments (A5.4) for two levels of trawling pressure (unfished, trawled 1x per year). Data: Bolam *et al.*, 2014; van Denderen *et al.*, 2015. Preliminary result from the BENTHIS project.

### 3.2.6 Possible improvements in assessing habitat sensitivity

Biological Traits Analysis (BTA) has a foundation in Habitat Template Theory (Southwood, 1977, 1988), which states that species' characteristics evolve in response to habitat giving characteristic life-history strategies. Community structure is governed by habitat variability and response to stressors such that the biological traits exhibited by organisms reflect environmental status (Usseglio-Polatera *et al.*, 2000), recognizing that the phylogenetic composition of the constituent taxa will constrain specific traits, independently of habitat (Gould and Lewontin, 1979).

The BENTHIS traits introduced in WGEKO 2015 can be applied to improve the approach based on mortality and recovery (Section 3.2.2) which is now based on the estimated sensitivity of the whole benthic community. The selection of a suite of benthic species with traits that are perceived to increase mortality or decrease the recovery potential should result in an improved estimate of the sensitivity of the benthic community and hence a more precautionary assessment of the state of the seabed and how this is affected by fishing. The effects of fishing impacts on physical habitats has been considered previously, and in general soft sediments are perceived to be less sensitive than hard substrata. However, this is a very generalized relationship that should be further examined on a case by case basis.

Table 3.3 presents trait modalities affecting mortality and/or recovery of benthic species. They can serve to guide sensitivity assessment (Step 4). The infauna are especially noted in order to select traits that reflect subsurface abrasion, recognizing that infauna are not the most susceptible component in the benthic community to some gear types. The modalities are also reflective of those found in the North Sea and could easily be expanded to include a broader range (for example size classes >500 mm could be produced) and new traits could be introduced to better reflect the species in the region of interest. Bremner *et al.* (2006) provided nine biological traits describing the life history, morphology and behaviour of megabenthic invertebrates which were perceived to be relevant to ecosystem process and also sensitive to envi-

ronmental disturbances. Of the eleven traits listed in Table 3.3, four are likely to reflect instantaneous impact, three are likely to reflect recovery trajectories and two have modalities that are correlated with both impact and recovery. Interestingly, longevity (Section 3.2.4) was not directly linked to either impact or recovery. Therefore the use of this single trait requires examination of correlative traits in order to validate the approach. This was done in the context of sensitive fish species (ToR C) and is recommended for benthic species. Bioturbators were also not directly linked to mortality or recovery, being a functional trait, however individual bioturbators, and bioturbating communities would likely be captured by a combination of the other traits.

**Table 3.3. Biological traits and modalities (WGECO, 2015) perceived to be associated with increased and decreased mortality and recovery following fishing disturbance.**

Trait	MODALITIES ASSOCIATED WITH			
	Increased mortality	Decreased mortality	Increased Recovery	Decreased Recovery
Morphology	Soft; Tunic; Stalked	Exoskeleton; Crustose; Cushion		
Sediment position	Surface ; Infauna: 0–5 cm	Infauna: 6–10 cm; Infauna: >10 cm		
Mobility	Sessile; Crawl/creep/ climb	Swim; Burrower		
Protection	No protection; Fragile	Tough skin or exoskeleton; Robust		
Feeding mode			Scavenger/opportunist; Predator	Parasite
Larval development			Pelagic – Planktotrophic; Pelagic – Lecithotrophic	Benthic– Direct
Egg development location			Asexual/budding; Sexual-shed eggs-pelagic	Sexual-shed eggs-benthic; Sexual-brood eggs
Living habitat	Tube-dwelling; Epi/endo-zoic/phytic;Attached	Burrow-dwelling; Crevices/holes/under stones	Free-living	Tube-dwelling; Epi/endo-zoic/phytic; Attached
Maximum body size (length) in mm	>500	<10, 11–20, 21–100, 101–200, 201–500	<10, 11–20, 21–100, 101–200, 201–500	>500

### 3.3 Discussion

The WKFBI terms of reference relate to a mapping approach to the development of the seabed integrity indicator for GES, specifically Descriptor 6.1. The Canadian case study also presents such an approach. However, that case study is fundamentally different from the North Sea case, as the sensitive species of an area are identified *a priori* and then mapped to identify distribution and interaction with fisheries (i.e. Steps 4 and 5). In this way it has more in common with the approach being considered for fish species (see ToR C).

Various modelling approaches were used towards identifying robust areas for assessment of fishing impacts (Step 5). The models used had different strengths and weakness. Some, such as the KDE mapping, use actual data locations with minimal interpolation. This can work well in areas that are well surveyed for the species under consideration, but in many areas the surveys have spatial gaps and in those instances SDM can be used to extrapolate to unsampled areas using environmental predictors, with the *proviso* that extrapolation outside the environmental data ranges occupied by the response variables is highly uncertain. This creates two classes of distribution maps: actual and potential, with the latter having varying degrees of extrapolation depending upon the input data. Management decisions in the Northwest Atlantic have thus far been taken from actual distributions with potential distributions being used to reinforce or refine the former. However this could change as fisheries develop in frontier areas. There SDM can help managers to assess the potential impact such fisheries could have on the environment before they are authorized.

The Canadian case study modelled groups of taxa with similar environmental preferences (similar to the “archetypes” of Hui *et al.* (2013)). They were modelling habitats created by structure-forming species which were considered vulnerable marine ecosystems. Knudby *et al.* (2014) also took this approach in modelling the distribution of sponge grounds in the Northwest Atlantic. For the approach to be applicable to the assessment of seabed integrity under GES, a range of “archetypes” (see Section 3.1.1.1) would have to be quantitatively determined and mapped (e.g. Foster *et al.*, 2014). This approach would allow community level assessments of impacts. As seabed type is often closely linked to physical oceanography and seabed topography, such predictors would likely cross-connect with the approach that links species impact and recovery to habitat class. However, different benthic communities have different levels of stability in terms of species composition with some species having lifespans well under a year. The vulnerable marine ecosystem indicators used in the Canadian example are typically very long-lived taxa and so maps of their distribution are expected to be relevant on decadal time-scales. Conversely, actual maps of communities dominated by short-lived species can be expected to have more temporal variability and maps of potential distribution based on environmental data may be more stable in those instances.

The seabed integrity indicator has thus far been considered under a community response. Therefore traits-based approaches require some harmonizing of the range of traits shown by a community as every benthic community will have mixtures of biological traits. It may be more desirable to identify the most sensitive species in any spatial unit and base the seabed integrity evaluation on them following a precautionary approach.

### 3.3.1 Relevance in the context of WKFBFI

The guidance provided for the WKFBFI distinguishes specific steps that are required for assessing how pressure maps of fishing intensity contribute to an assessment of the state of seabed habitats and how these are impacted by fishing. Below we consider the relevance to each of these steps of the different approaches considered for this process and presented in Section 3.2.

- 1) Acquire a habitat map covering as much of the MSFD region as possible. The thematic classes of the habitat map need to be aligned or cross-referenced to the classes used in the sensitivity assessment without significant gaps.

Habitat maps available to WKFBFI are maps of physical habitats as discussed above. In contrast, modelling approaches used in the Canadian case study map biogenic habitats that were *a priori* identified as sensitive. These modelling approaches could be extended to map other biological communities in the MSFD process. Using only physical habitat maps for this process assumes that they stand as good proxies for biological communities which may not always be the case. In fact, many benthic species respond only coarsely to physical habitat types (hard or soft sediments for example) and the many habitat types portrayed in the physical maps may not be reflective of biological heterogeneity. The mortality/recovery approach attempted to cover some of this heterogeneity by introducing the variation in recovery potential and carrying capacity that are found within natural communities.

Consequently, while the impact assessment (in steps 4–6) using these habitat maps will provide information beyond what the pressure maps can provide, they would benefit from knowledge of the actual small-spatial-scale composition of the benthic community possibly combined with additional information on the sensitivity of the benthic community as represented by the traits such as presented in Section 3.2.6.

- 2) Acquire sensitivity information for each thematic class of habitat to surface/subsurface abrasion. Habitat map polygons are then attributed with the sensitivity code.

The available sensitivity information differs in terms of:

- Number of habitat categories. The expert judgement-based scoring method is the most detailed and comprehensive distinguishing between surface and subsurface. The other approaches cover less detailed categories.
- Type of information differing between qualitative scores based on expert judgement and quantitative scores based on an analysis of fishing impact experiments vs. the occurrence of a life-history trait.

The scoring process has the advantage that it includes and distinguishes all the occurring habitats in their scoring but the categories are very crude (H, M, L) and it is unclear, with the information provided and how these categories are supposed to be combined with fishing pressure (determined by intensity and type of gear) to derive an impact.

The application of both the mortality/recovery and longevity methods to the wider MSFD assessment area is restricted by the habitats and benthic species for which appropriate information is available. The parameters of the mortality/recovery method are based on a global meta-analysis of shelf-sea benthic

communities resulting in four fairly coarse habitat categories (i.e. Sand, Mud, Gravel and Biogenic). These categories then need to be linked to the more detailed lower level OSPAR habitat categories occurring in the habitat maps available for each of the MSFD (sub)regions on which sensitivity will be evaluated. The same applies for the longevity method where the longevity distribution is only available for a limited number of habitats.

- 3) Acquire surface/subsurface abrasion layers (pressures from fishing activity) for the MSFD assessment area. Clip layers to match the habitat map coverage.

Only the scoring method distinguishes between surface and subsurface abrasion. In time it should be possible to derive sensitivity parameters for surface and subsurface benthos separately.

- 4) Combine (intersect/raster calculator/map algebra) the attributed habitat map with the abrasion layers. Use a combination matrix (categorical attribution of pressure and sensitivity) to combine sensitivity and pressure to calculate impact.

The combination of the scoring method with the available pressure maps can only result in categorical impact maps unless a method exists that allows a translation of the qualitative scores into sensitivity values that can be combined with the pressure information into impact estimates. To our knowledge such a method is not (yet?) available.

Both the mortality/recovery and longevity methods can create quantitative impact maps but differ in terms of their calculation of seabed integrity and how this is impacted by fishing. An advantage of the mortality/recovery method over the longevity method is that it better captures the dynamics over time because it explicitly considers the state of the seabed (affected by historic fishing activities prior to the assessment year) when calculating impact. This is possible because the recovery component is explicitly parametrised in the mortality/recovery method while the longevity method calculates the seabed integrity every year without any consideration of the past exploitation. Notably for habitats with slow recovery rates (>1 year) and high historic impacts the longevity method will therefore systematically overestimate impact.

Another advantage of the mortality/recovery method is that sensitivity is gear- and habitat-specific while in the longevity method sensitivity appears to be only habitat-specific. The disadvantage of the mortality/recovery method is that it assigns a single sensitivity estimate for each of the four substrate types, based on meta-analysis of data from fishing impact studies. If we assume that the meta-analysis shows that the average loss of biomass is 30% on muddy seabeds, then assigning this value to muddy seabeds in all locations poses a risk. The sensitivity of a benthic communities found on a particular substrate type can differ from place to place. As an example, fishing may cause 30% decrease in biomass on muddy substrate in one location but 90% in another location. One approach would be to predict distributions of benthic habitats or communities, and examine the sensitivity of these habitats to fishing impacts.

- 5) Map the impact of fishing on benthic habitat

The different methods can calculate impact at different levels of accuracy for grid cells determined by the quality of the habitat and fishing pressure maps. How this should be mapped was not considered by WGEKO.

- 6) Extract summary statistics/indicators from the impact map. Produce a confidence assessment for the map and summary statistics.

Summary statistics can be easily derived from the maps by integrating the information in the grid cells into annual (sub)regional estimates of seabed integrity. The difference in seabed integrity between two subsequent years can then be related to fishing impact in relation to habitat sensitivity (which includes recovery). In theory this can result in negative impacts implying a recovery of the seabed.

Some further considerations on the suitability of the available methods to be used as part of an impact assessment as well as to inform ecosystem-based management (EBM) are required if the summary statistics show that policy objectives were not achieved. The advantage of the mortality/recovery is that it is firmly embedded in the DPSIR framework allowing this method to be applied not just to create impact maps based on seabed integrity but also to determine the impact caused by specific fishing activities (i.e. fishing métiers or even per trip) over time and as such it can inform EBM. The damage caused by any specific subset of the fishery (e.g. métier or trip) can be estimated more accurately with the mortality/recovery method since:

- It uses the actual (i.e. annual) sensitivity also including the effect of previous fishing activities (i.e. up to the beginning of each year that is to be assessed).
- The parameters used to determine impact are habitat- and gear-specific (i.e. distinguishing beam trawl, otter trawl and dredge). These categories will need to be linked to the (more detailed) métiers occurring in each region.

An advantage of the mortality/recovery approach for assessment purposes is that this method proposes potential boundaries for GES based on the conclusion in (Rice *et al.*, 2012) that GES cannot be defined exclusively as “pristine Environmental Status” but rather as the status when impacts of all uses are sustainable. This results in two boundaries which capture those perspectives and are aligned with the two GES criteria coming from the (EC, 2006) criterion that there should be “no significant deterioration in quality or pressures threatening” where the first part, i.e. “no significant deterioration in quality”, links to a GES boundary requiring an SI that does not differ significantly from an unfished situation (the  $x\%B_0$  in Figure 3.5), while the part referring to pressure levels links to a GES boundary which requires “the level of fishing disturbance to be below the recovery capacity of the benthic community”. Figure 3.6 shows how these two criteria are related because if applied consistently over the long term, a specific pressure (i.e. type of gear fished at a specific trawling frequency) will result in a specific deterioration in quality (dependent on the sensitivity of the habitat). Table 3.2 shows how the concept of significant deterioration (effectively allowing recovery to an  $x\%$  of unimpacted biomass  $B_0$ , e.g.  $80\%B_0$ ,  $90\%B_0$  or  $95\%B_0$ ), determines which trawling frequencies can be considered impacted.

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#### **4 Investigate possible indicators of scavengers, examine their relation to discard amounts and evaluate the spatial effect of a landing obligation on the scavengers (ToR b)**

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In 2015, WGECO set out to investigate possible relationships between benthic scavenger species and discarding. Three activity areas were defined:

- 1) Activity 1 – Investigate possible indicators for proportion of benthic biomass which are key scavengers. A term of reference will be constructed in collaboration with the Benthos Ecology Working Group (BEWG).
- 2) Activity 2 – Investigate possible indicators for biomass or abundance of key scavengers. A term of reference will be constructed in collaboration with BEWG.
- 3) Activity 3 – Investigate possible indicators based on important areas of spatial overlap of key scavenger species distribution and discards from main fleets.

WGECO brought together the following data to investigate under these activities:

- Discard biomass data for the North Sea derived from the STECF database and developed for the DiscardLess project atlas 2013 and 2014.
- Scavenger abundance distribution data from the database for epibenthic species abundance developed by the BENTHIS project.
- Swept-area estimates by gear type for the North Sea developed by the BENTHIS project (Eigaard *et al.*, 2015).
- Probability distribution data for hagfish from FishBase.

These data were made available prior to the meeting. Other relevant databases were identified during the meeting, and will be worked on intersessionally:

- Data on hagfish (and possibly other benthic species) abundance distribution from the MAFCONS project.
- Spatially resolved data on benthic biomass made available to scavengers from trawl path mortality from the BENTHIS project.

The analysis is ongoing and will be reported briefly here, aiming at a complete report in 2017.

Initial examinations of the data (Figures 4.1 and 4.2) showed that the choice of species identified as “scavengers” was critical in developing scavenger abundance distribution maps. WGECO used the groups identified in the BENTHIS database, and a different species choice based on the list developed by WGECO in 2015, and agreed with BEWG. These showed radically different distributions both in terms of abundance and of scavengers as a proportion of the full epibenthic community. In particular, the first showed the main abundance in the southern North Sea and the second with the main abundance in the Northern North Sea.

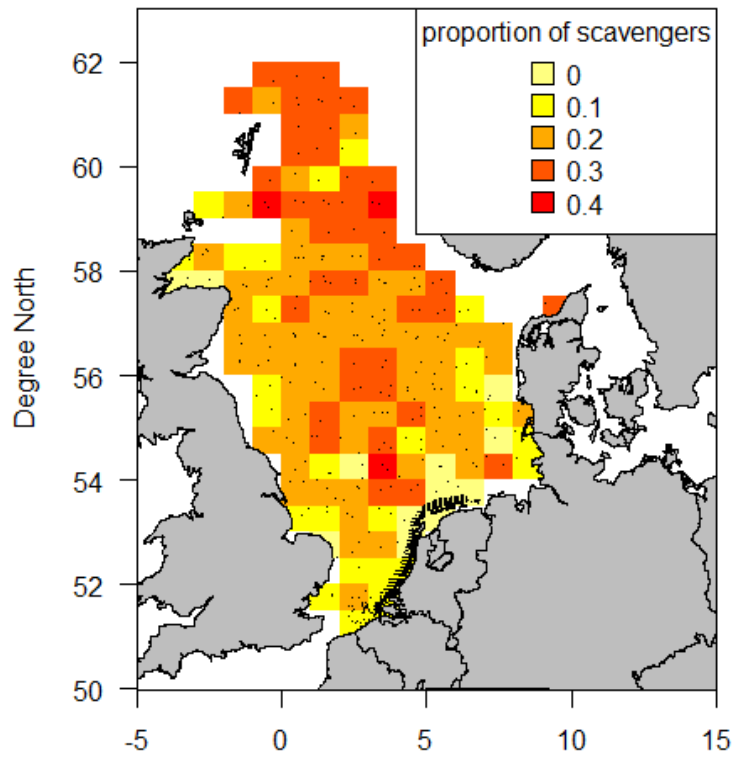


Figure 4.1. Scavenger abundance as a proportion of the total benthic community, based on the BENTHIS selection and full BENTHIS database (Bolam *et al.*, 2014a, b).

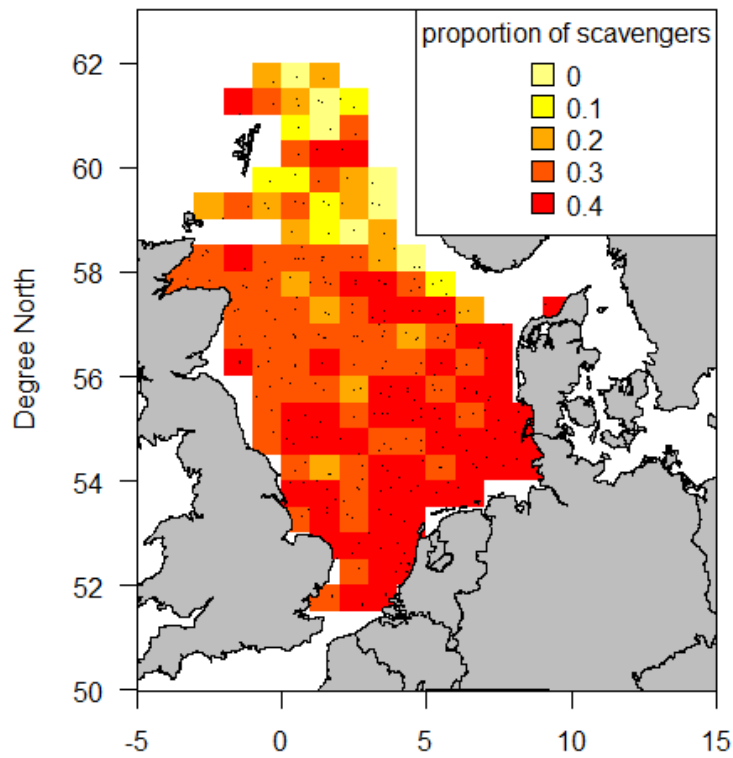


Figure 4.2. Scavenger abundance as a proportion of the total benthic community, based on the WGECO selection and the UK data in the BENTHIS database (Bolam *et al.*, 2014a, b).

Given the knowledge of the differences in oceanography, fish community, dominant fishing gears and the benthic community, it was agreed to separate the analysis into two components for the southern and northern North Sea. The dividing line was taken from Fraser *et al.* (2008). Given that much of the data on scavengers used by WGECO in 2015 (ICES, 2015) was derived from the southern North Sea, the 2016 analysis focused on the main southern species (by abundance), and derived from the Cefas beam trawl data within the BENTHIS database (Bolam *et al.*, 2014a). At the time of working, these data were the only species disaggregated benthic data that were available to WGECO, but cover the entire area in question.

To date, all analysis has been carried out at an ICES rectangle scale, as this is the scale on which discard data for all countries were available from the STECF database.

#### 4.1 Southern North Sea

Figure 3 shows the relative total abundance by taxa for the benthic community in the southern NS derived from the Cefas data. Based on the WGECO 2015 list of main scavengers, the analysis focused on the most abundant of those taxa from Figure 4.3 (*Anapagurus sp.*, *Asterias sp.*, *Astropecten sp.*, *Crangon sp.*, *Liocarinus sp.*, *Macropodia sp.*, *Ophiura sp.*, *Pagurus sp.*).

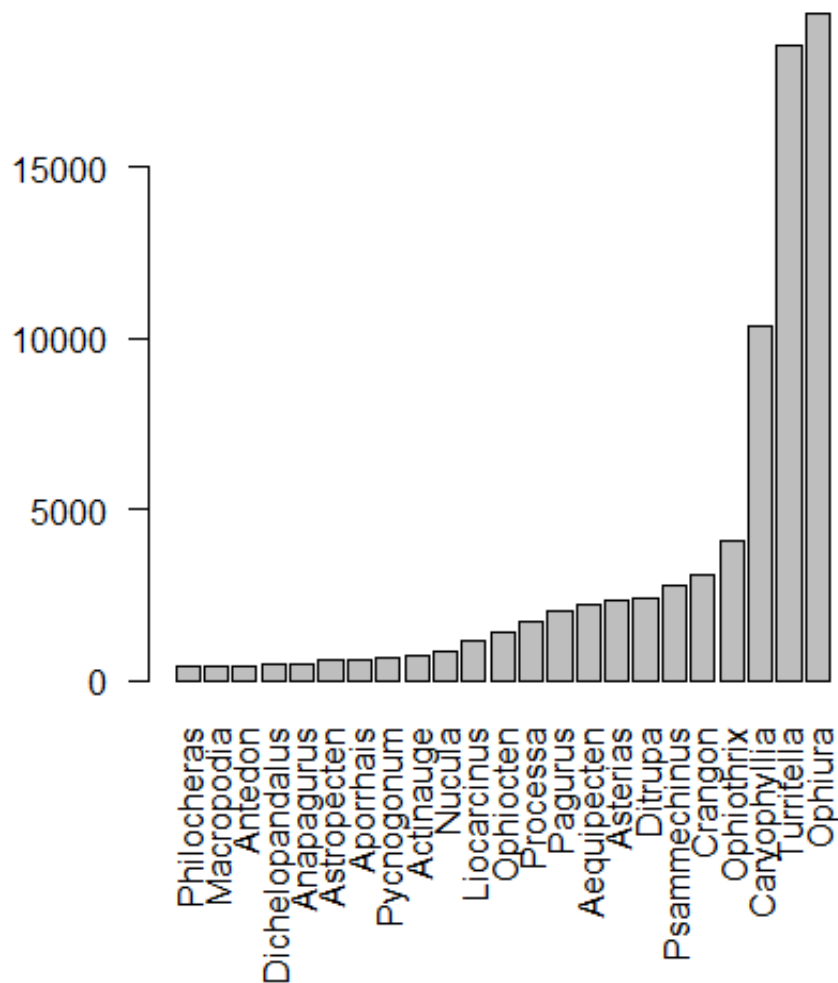


Figure 4.3. Abundance by taxa across the southern NS from the Cefas beam trawl survey database.

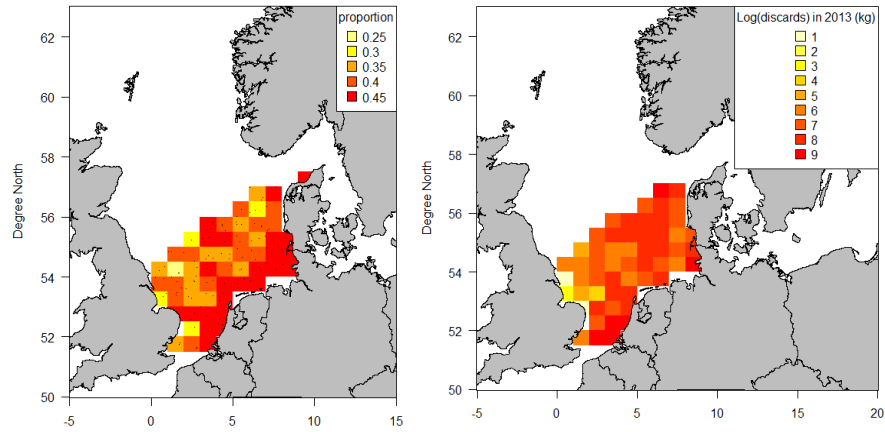


Figure 4.4. Scavenger abundance as a proportion of the total benthic community, based on the WGECO selection (left), and logged discard biomass for 2013 (right).

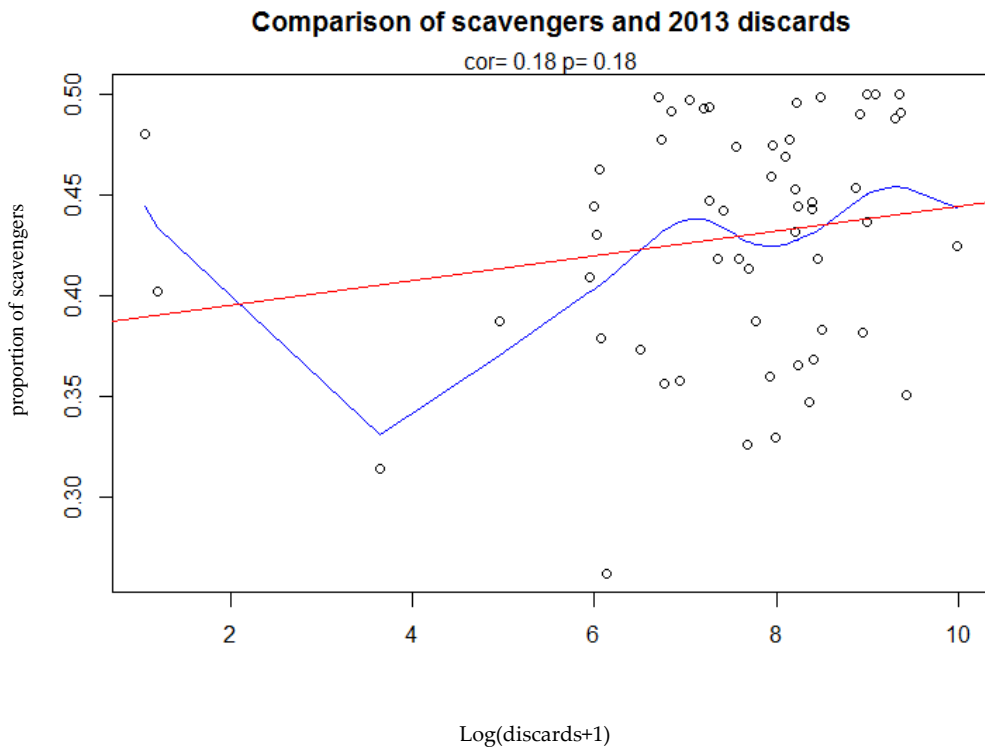


Figure 4.5. Scatterplot of the data from Figure 4.

The relationships between discard biomass and scavengers as a proportion of the benthic community, is presented in Figures 4.4 and 4.5. There was a slight, but insignificant positive relationship.

We also looked at individual “scavenger” species. One example for *Pagurus* spp. (hermit crabs) is presented below in Figures 4.6 and 4.7, again, there was a slight, but insignificant positive relationship.

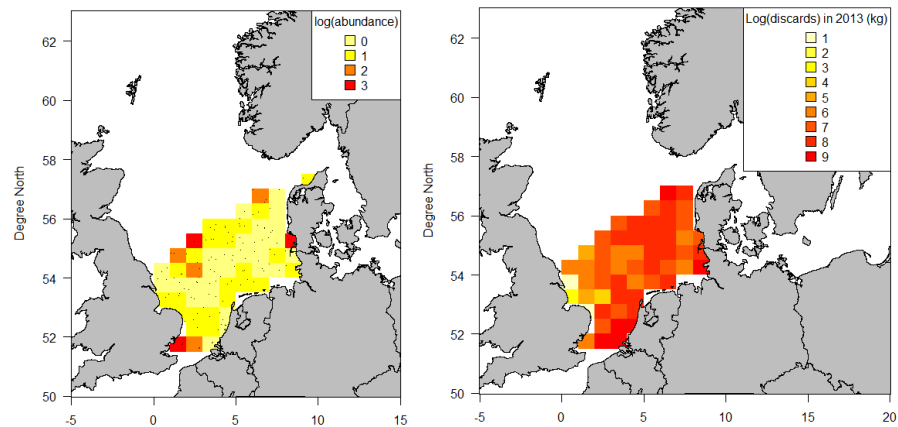


Figure 4.6. *Pagurus* spp. abundance from the Cefas database (left), and logged discard biomass for 2013 (right).

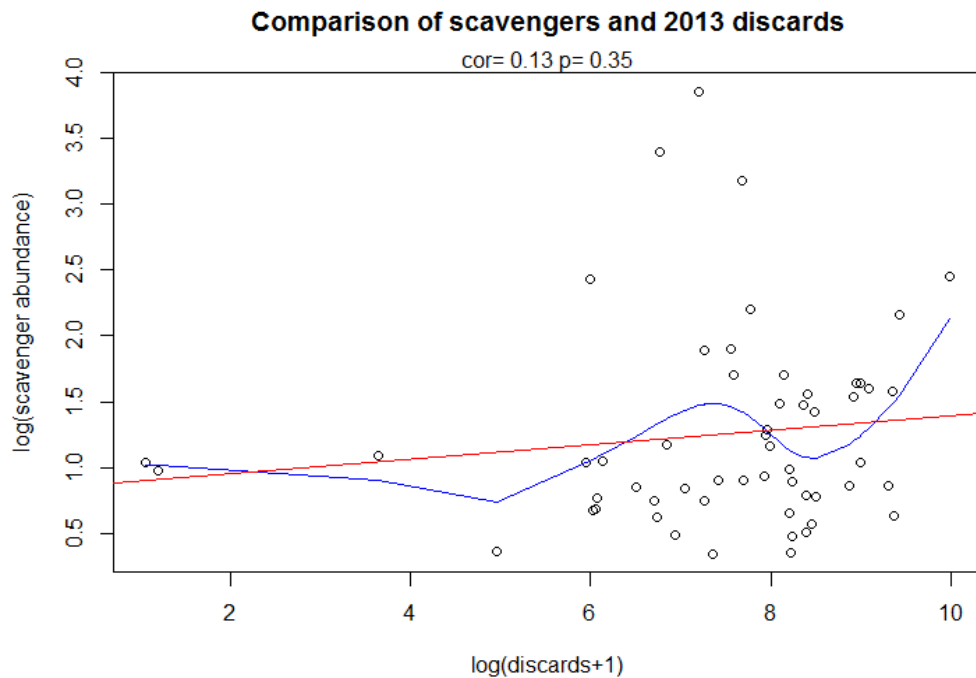


Figure 4.7. Scatterplot of the data from Figure 6.

Similar non-significant relationships were found for other prominent “scavenger” species.

This analysis was carried out at an ICES rectangle scale, and at this scale, there was no evidence of any link between scavenger abundance and discarded biomass. It is possible, however, that at this spatial scale, it may be difficult to detect such a link. The next stage in the analysis will focus on the station based data from the Cefas beam trawl dataset. WGECO proposes to partition the discards from the STECF database into smaller cells in 2017 (possibly 5 x 5 nm cells within each rectangle) to repeat the analysis using the station scavenger data with the local discard biomass. Partitioning will make use of the database on swept-area by gear developed for the BENTHIS project (Eigaard *et al.*, 2015), and the discards by gear from the STECF database.



### 4.2 Northern North Sea

Based on the FishBase modelled distributions of hagfish, and also from baited camera work at Marine Scotland Science and DTU-Aqua, hagfish are a key scavenger species in the north. The analysis therefore focused on hagfish in the northern North Sea.

Figure 4.8 shows the total discard biomass for 2013 and 2014 for the northern NS. Figure 4.9 shows the modelled probability of finding hagfish from the FishBase modelled data. Figure 4.10 shows the scatterplots and regressions for the two years. Both show insignificant positive relationships.

Similar to the southern NS, this analysis was carried out at the rectangle scale, and an examination at a finer spatial scale may prove more effective. Station scale data for hagfish abundance should be available from data collected during the MAFCONS project using 2 m beam trawls. WGECO propose to use this along with the finer scale discard data (described above) to explore the matter further in 2017.

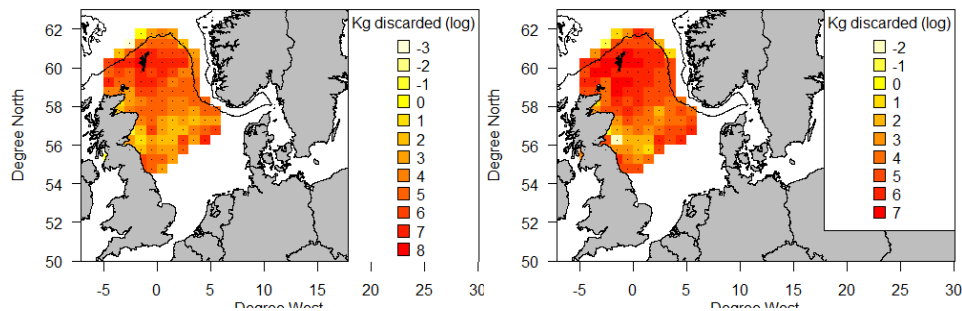


Figure 4.8. Log discard biomass by rectangle for 2013 (left) and 2014 (right) from the STECF database.

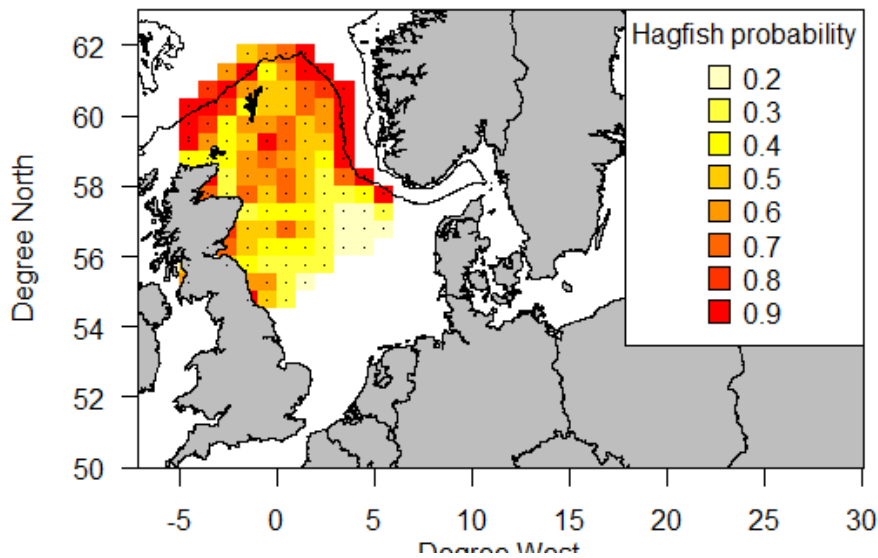


Figure 4.9. Probability distribution for hagfish based on the FishBase model.

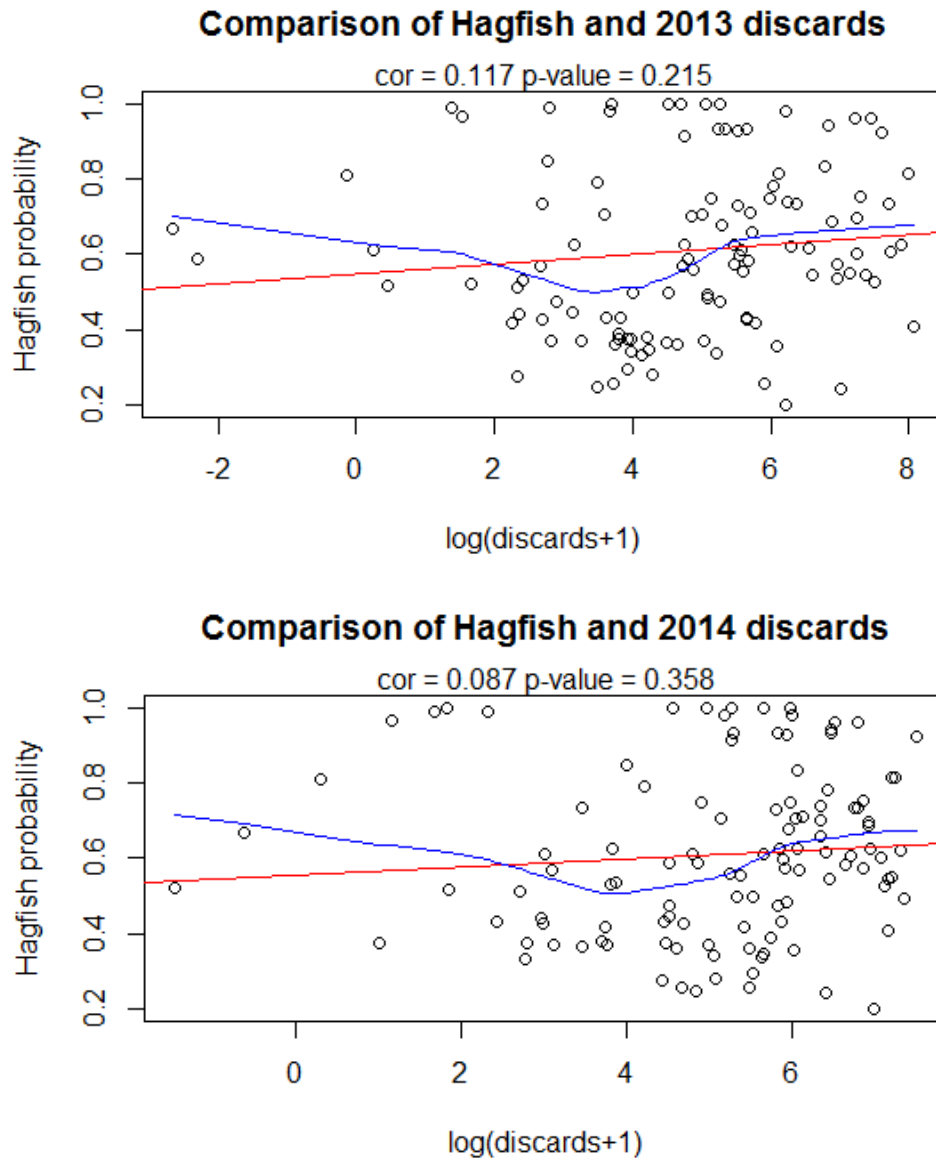


Figure 4.10. Scatterplot of hagfish probability and log discard biomass by rectangle for 2013 (top) and 2014 (bottom).

### 4.3 Conclusions and plans for progress

The broad conclusion from the analysis was that there was no demonstrable link between discard biomass and either scavengers' as a proportion of the benthic community or individual "scavenger" species abundance. It was also concluded that the analysis might be more successful at a finer scale of resolution than at the level of the ICES rectangle.

Therefore, the proposed future work would be to develop more detailed discard biomass maps, and use these for an analysis of the station based benthic abundance and ideally, if available, biomass data. This would essentially repeat the rectangle based analysis at finer scale.

A second major potential influence on scavenger abundance linked to fishing that was considered at the meeting, was carrion made available as a result of trawl track mortality of both epifauna and infauna. If this was substantially greater than the possible “subsidy” from discarding, one would not expect to find strong links to discarding. A short review of the literature (e.g. Hiddink *et al.*, 2006; Kaiser *et al.*, 2006; Kaiser and Hiddink, 2007; Enever *et al.*, 2014; Quirijns and Pastoors, 2014) suggests that in very general terms, discard biomass is likely of a similar order of magnitude to the biomass available from trawl track mortality. A dataset is available from the BENTHIS project that can provide a map of relative trawl track mortality, and this may be available in biomass terms in the near future. WGECO would therefore propose that the station based analysis should be carried out using:

- A more detailed spatial scale map of discards derived from the STECF database, partitioned using the BENTHIS swept-area by gear database.
- The station based benthos species resolved abundance (and ideally biomass) data from the BENTHIS database.
- The BENTHIS database on trawl track mortality biomass.

This could be done initially for the southern North Sea and, depending on the outcome, the northern North Sea.

For the Northern NS, WGECO will use that same more highly resolved datasets described above, but start with an analysis of any possible links between hagfish abundance (biomass) and the more spatially detailed discard and trawl track mortality data in 2017.

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## 5 Use the data available to evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced” (ToR c)

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### 5.1 Examine time-trends in stocks that are rebuilding and dependent stocks

This ToR component continues work started by WGEKO in 2014 under ToR f (WGEKO, 2014). The background to this ToR arose from concerns that seabird and other dependent predator populations could be impacted from a combination of a landing obligation and MSY fishing targets for predatory fish. Seabirds depend heavily on forage fish species such as sandeel. Forage fish are important as prey for many predatory fish, and if these stocks increase, this has been suggested as a cause of forage fish decline based on both ecosystem modelling and analyses of dataseries. Seabirds are also known to make use of discards, which are expected to diminish following the landing obligation. Thus the possibility exists that seabirds in particular may face two threats to their food supply almost simultaneously. While this ToR was inspired by this particular case, the issue probably has relevance for many predator-prey interactions as commercial stocks move towards MSY targets. In 2014 WGEKO reviewed the evidence to support the generality of suggested effects of rebuilding predator stocks. Following this review, WGEKO recommended, among other priority areas for future research, a structured meta-analysis of the incidence of top-down predator control in fish communities (and for seabirds) which includes null results to avoid reporting bias. The analyses performed in 2016 follow from this recommendation.

The objective is to empirically examine time-trends in stocks that are rebuilding and their dependent stocks (prey and predators), and to compare these trends with expectations from simple ecological theory.

#### 5.1.1 Methods

Species were identified that had sustained increases in biomass over at least an eight year period. The group focused on predator species, for which trends in the abundance of their prey could be estimated. The period over which there was a sustained increase was defined as the interval between the minimum (just before the increase started) to the year of maximum biomass.

For each of the rebuilding species, the group identified the important prey species from diet analyses. Trends in prey species biomass, during the rebuilding period, were determined either from stock assessments or from survey data (Table 5.2.1.1). The trends in prey species biomass were tested in two ways:

- i) Rank correlation of prey-species biomass with time, using a 1-tailed probability to test for a negative correlation (decline in prey biomass).
- ii) Correlation of prey species biomass against rebuilding predator biomass (rank correlation or linear regression).

The temporal trends in prey biomass were interpreted with respect to predator abundance, exploitation rate, and recruitment dynamics. For example, if the prey species declined, did it start to decline prior to the predator rebuilding, which would suggest

some other cause? If there was an increase in prey, is it linked to a general decrease in fishing mortality, or common recruitment pattern, indicating bottom-up control?

### 5.1.2 Results

Six stocks were identified that have experienced sustained increases in biomass over periods of eight to 20 years (Figure 5.1). These rebuilding periods were long enough and the amount of biomass increase large enough that effects on dependent prey species could be expected.

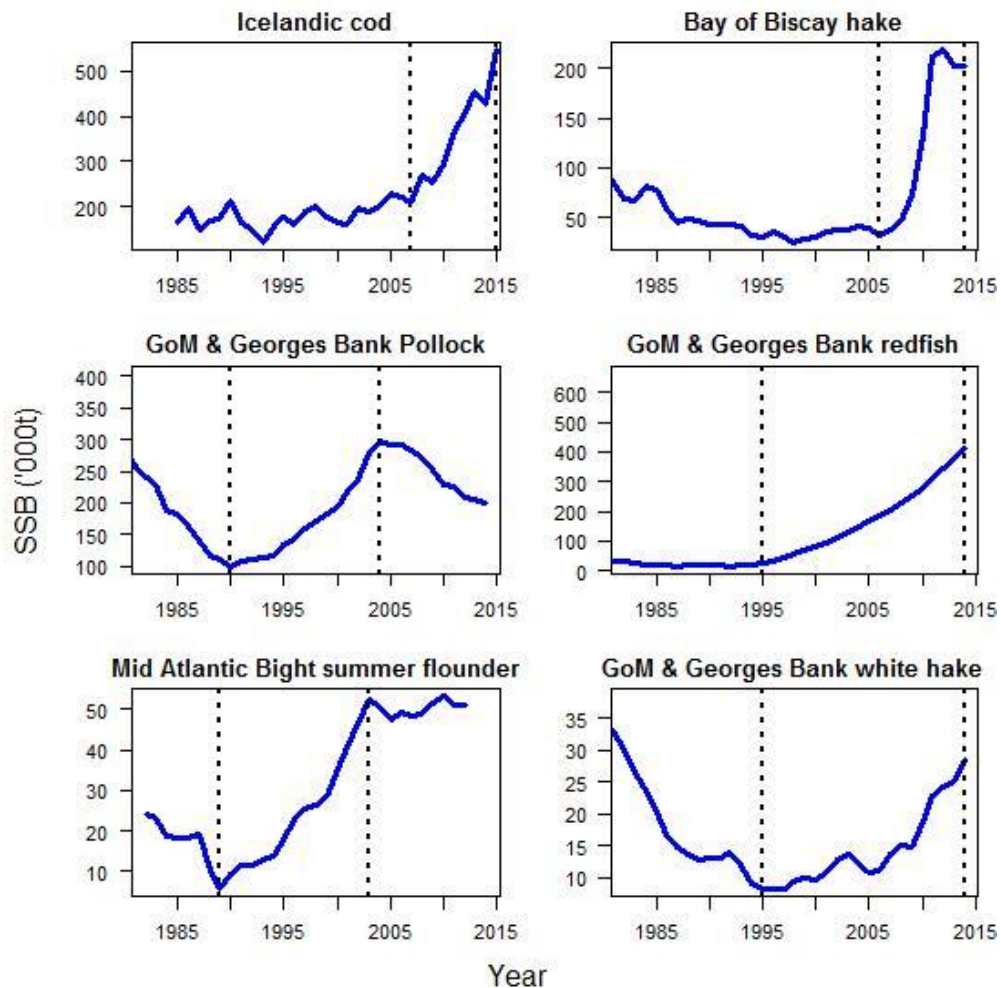


Figure 5.1. Time-series of Spawning-Stock Biomass (SSB) of predators analysed in this section. Dotted lines show the limits of the time period used for analysis.

Based on diet studies, one to seven dependent prey species were identified for each of the predators, for a total of 20 prey species (Table 5.1). Rank correlations between prey abundance and time and between prey abundance and predator abundance were both negative and positive. For most of the prey species the signs and probability levels of the two tests were consistent. Note that in the case of monotonic increases in biomass, rank correlations with time or biomass give identical results. For five of the predator-prey pairs, the correlations were negative and significant: Bay of Biscay hake feeding on horse mackerel, Gulf of Maine/Georges Bank hake feeding on herring (time-trend in herring only), and Mid-Atlantic Bight summer flounder feed-

ing on butterfish ( $p=0.05$ ), and Iceland cod feeding on herring and shrimp. These results are interpreted by region in the following sections.

**Table 5.1. Results of correlation analyses for (i) time-trends in prey biomass and (ii) correlation between predator and prey biomass. TSB Total-stock biomass; SSB Spawning-Stock Biomass; Fall B index of biomass from fall trawl survey; Spring\_B index of biomass from spring trawl survey.  $\rho$  Spearman rank correlation coefficient, P-value of one-sided test of  $H_0: \rho = 0$  (alternative:  $\rho < 0$ ).**

REBUILT PREDATOR	PREY STOCK	INDEX	(i) TIME-TREND		(ii) PREDATOR-PREY	
			$\rho$	P-value	$\rho$	P-value
Bay of Biscay hake	Horse mackerel	TSB	-0.83	0.004	-0.70	0.02
	Blue whiting	SSB	-0.28	0.23	-0.52	0.08
	Sardine	TSB	0.22	0.72	0.08	0.60
	Anchovy	SSB	0.78	0.99	0.92	1.00
	Bib	Fall_B	0.37	0.84	0.07	0.58
	Poor cod	Fall_B	0.18	0.69	0.28	0.78
Gulf of Maine and Georges Bank Pollock	Sandeel	Spring_B	0.48	0.96	0.48	0.96
	Herring	Spring_B	0.09	0.63	0.09	0.63
	Butterfish	Fall_B	-0.38	0.08	-0.38	0.08
	Silver hake	Fall_B	-0.21	0.22	-0.21	0.22
Gulf of Maine and Georges Bank White hake	Herring	Spring_B	-0.44	0.02	-0.31	0.07
	Silver hake	Fall_B	0.24	0.87	0.38	0.96
Mid-Atlantic Bight Summer flounder	Sandeel	Spring_B	0.15	0.71	0.15	0.71
	Herring	Spring_B	0.31	0.88	0.31	0.88
	Mackerel	Spring_B	0.25	0.82	0.25	0.82
	Butterfish	Fall_B	-0.45	0.05	-0.45	0.05
	Scup	Fall_B	0.10	0.64	0.10	0.64
	Silver hake	Fall_B	-0.07	0.41	-0.07	0.41
	Squid	Fall_B	0.06	0.59	0.06	0.59
Gulf of Maine and Georges Bank Redfish	Silver hake	Fall_B	0.30	0.90	0.30	0.90
Cod in Icelandic waters	Capelin	Biomass	0.23	0.74	0.47	0.90
	Herring	Biomass	-0.85	0.009	-0.75	0.01
	Shrimp	Biomass	-0.92	0.001	-0.85	0.003

#### 5.1.2.1 Bay of Biscay

The Northern stock of European hake rebuilt rapidly after the implementation of the rebuilding plan adopted in 2004 (European Union, 2004). Fishing mortality decreased as soon as the rebuilding plan was launched, and Stock Spawning–Stock biomass increased by almost a factor of ten compared to its minimum 1998 level. Among the six most abundant prey of this species, only horse mackerel had a declining trend concomitant with the increase in hake. However, horse mackerel is a wide ranging stock spanning a large part of the North Atlantic area; and the decline in horse mackerel coincided with an increase in fishing mortality on this stock (ICES Advice, 2015). Thus it is not likely that decline in horse mackerel can be ascribed to an increase in predation by hake.

#### 5.1.2.2 Iceland

The cod stock in Icelandic waters has been steadily increasing since 2007, in response to reduced fishing effort (Figure 5.2). A larger cod stock may, however, influence the size of its main prey stocks: capelin and northern shrimp. Based on stomach-content analysis in the annual spring survey, summer-spawning herring is also of importance in some years. The mean weight-at-age of cod age classes 3–11 has been increasing in the period. The weight-at-age for the older age classes vary, probably due to low sample sizes.

Since 2007, the shrimp and summer-spawning herring indices have been steadily decreasing (Figure 5.2). In the same period, the cod exploitation (landings / biomass index) has decreased while exploitation increased for shrimp due to an increasing number of boats in the shrimp fishery. The exploitation rate for summer-spawning herring and capelin has fluctuated without trend during this period. Recruitment is fluctuating without any trend in all species. The changes in biomass trend can therefore not be explained by recruitment.

However, increasing exploitation may have had a negative effect on the shrimp stock. Increased predation pressure by cod on shrimp is, however, likely to have a greater effect on the shrimp stock in Icelandic waters due to the close linkage between shrimp and cod. Several studies have noted the importance of predatory control of shrimp by cod (Worm and Myers, 2003; Wieland and Siegstad, 2012; Jónsdóttir *et al.*, 2012) resulting in opposite trends of decreasing shrimp biomass and increasing biomass of cod (Anderson and Piatt, 1999; Mueter and Norcross, 2000) or vice versa (Worm and Myers, 2003).

The rebuilding of the cod stock does not seem to influence the capelin biomass during this time period. Capelin is the most important prey of cod in Icelandic waters, and a clear bottom-up effect has been reported (Pálsson and Björnsson, 2011). The herring stock was heavily infected by *Ichthyophonus* during the period studied here, which caused increased natural mortality in this stock. In summary, there is support for the hypothesis that stock rebuilding affects dependent prey species for two of three prey species of cod in Icelandic waters but the relative importance of predator rebuilding and other factors are difficult to determine.

#### 5.1.2.3 New England

Of the ten prey species considered in the New England region, two exhibited declines that were consistent with predator rebuilding. The weight-at-age of white hake in the Gulf of Maine and Georges Bank varied without trend during the rebuilding period



1995–2014. Atlantic herring is an important prey species of white hake and other piscivorous fish (Smith and Link, 2010). Fishing mortality of herring was low and steady during the period 1995–2014 (Deroba, 2015). The stock assessment model included a 50% increase in the natural mortality rate starting in 1995. This increase was justified by estimates of the total consumption of herring by piscivorous fish, marine mammals and seabirds. Thus the declines in herring biomass indicated by the survey data likely result from aggregate predation by white hake, pollock, redfish, and other predators.

Weight-at-age of summer flounder has declined from 1989 to present, which is consistent with the hypotheses of food limitation as the stock rebuilt. Butterfish is preyed on by summer flounder but only constitutes 3% of its diet (Smith and Link, 2010). Summer flounder is one of six principle predators of butterfish (NEFSC, 2014). Time-series analysis of the total consumption of butterfish supported the use of a constant natural mortality in the assessment model, which suggests that predation mortality has not increased. The assessment model results imply that fishing mortality has declined, but it has always been low relative to natural mortality. Stock size has varied over the timespan of the assessment model, but has increased in recent years. No strong trend in recruitment was indicated over the timespan (NEFSC, 2014).

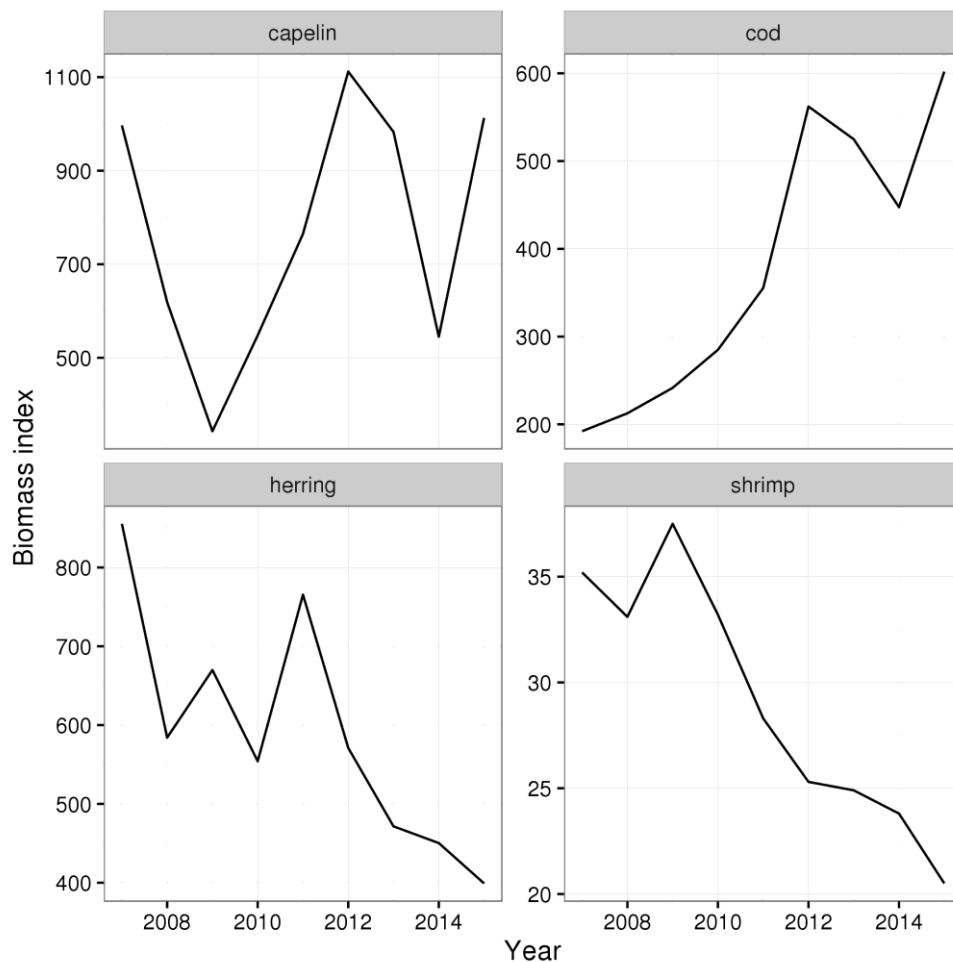


Figure 5.2. Time-series of four fish species in Icelandic waters.

### 5.1.3 Conclusions

For the six predator species considered, spawning-stock biomass increased by factors of 3 to 10, in most cases to levels similar to those prior to the stock declines. Thus, these recoveries might be expected to have significant trophic effects. Five of 20 prey species identified exhibited declines that could be significantly related to rebuilding of their predators. For only one species, shrimp in Icelandic waters, was there a strong case for linking the decline to increasing predation pressure. In the other ecosystems there are many weak predator–prey linkages between predator and prey species. Each predator has many prey species; each prey species has many predators. As a result, the increased trophic demand of rebuilding stocks may be attenuated in the foodweb. If so, rebuilding plans for individual species may not have immediate observable effects on dependent species. Impacts on dependent forage species could occur at local scales or if many predator stocks are rebuilt simultaneously, as may be the case for Atlantic herring in New England.

### 5.1.4 Future work

This empirical approach simply tests if there are observable declines in dependent species as predator stocks rebuild. Multispecies or ecosystem models could be used to make more specific, qualitative or quantitative predictions of the expected consequences of stock rebuilding. Without additional information, it is difficult to distinguish the different processes affecting prey species populations (predation, fishing, recruitment dynamics, climate induced distributional shifts). Food habits data can be combined with stock assessments to estimate the predation mortality on each prey species. Direct estimates of predation mortality would help to interpret the trends in prey species abundance. Finally, multispecies models that have been parameterized for the different areas can reconstruct the abundance of predator and prey species, providing estimates of fishing and predation mortality.

## 5.2 Establish the distribution of total catch among species (catch species dominance curves)

This ToR component continues work started by WGECO in 2014 under ToR c. The purpose of this ToR in 2014 was to consider whether all the likely impacts of a shift to balanced fishing would be potentially beneficial. Although metrics describing the selectivity or concentration of fishing across ecosystem components have been developed, WGECO 2014 found little empirical evidence that could be used to evaluate the ecosystem effects of balanced fishing. Existing studies provide scarce evidence that fishing patterns would directly determine community structure and biodiversity. Exploitation patterns may affect the community responses to changes in the environment or fishing intensity, but the size of these effects is not such that a strong signal can be detected among the noise of the many other factors. Consequently, WGECO recommended that broader-scale analyses of the actual fishing regimes would be useful. The work conducted by WGECO in 2016 responds to the 2014 recommendation by estimating the degree of fishing balance in a number of ICES regions. During this meeting, the group concentrated on the distribution of catch among species because size data were not readily available.

### 5.2.1 Methods

Landings data were extracted from national and ICES databases (Table 5.2.1.1). In each area, a five to ten year window was chosen during which there were no major

changes in the fishery or ecosystem. Unusual species (e.g. mussels, algae) that would not be caught with survey gear were removed. Other invertebrates (e.g. shrimps, crabs) were retained if they are also recorded in the corresponding survey. For the New England data, a number of landings records were omitted because of species codes that were not contained in the species names table. The landings corresponding to these missing species codes were minor and therefore would not affect the results reported here.

Landings of the retained species were expressed as proportions of their sum. The subset of species accounting for 95% of landing was identified. Evenness of the species distribution in the landings was calculated according to Simson's reciprocal index, corrected for the number of species  $N$ :

$$\frac{1}{\sum_i l_i n}$$

where  $l_i$  is the proportion of species  $i$  in the landings.

**Table 5.2. Sources of data used to calculate the distribution of species in landings and surveys.**

AREA	ASSESSMENTS	LANDINGS (YEARS)	SURVEY (YEARS)
Bay of Biscay	ICES (2015)	ICES (2009–2014)	WIBTS (2009–2014)
North Sea	ICES (2015)	ICES (2006–2014)	NSIBTS (2006–2015)
Iceland	MRI (2015)	Directorate of Fisheries	MRI (2003–2007)
New England	NEFMC (2015)	NEFSC1 (2000–2009)	NEFSC1 (2000–2009)

<sup>1</sup> Northeast Fisheries Science Center. Sean Lucey (personal communication).

Survey biomass data were extracted from national and ICES databases (Table 5.2). The survey data for New England only were adjusted for the catchability of species in the survey trawl, based on a set of coefficients derived by the NEFSC. Although it would be appropriate to adjust for catchability in the other areas as well, catchability coefficients could not be made available at the appropriate scale during the meeting. After initial screening for well-represented species, survey biomass was converted to proportions of the total survey biomass. The distributions of species both in the landings and surveys were expressed as proportions of their respective totals:

$$l_i = \frac{L_i}{\sum_j L_j} \text{ and } b_i = \frac{B_i}{\sum_j B_j}$$

and plotted with pie charts and bar plots.

Landings were compared with survey biomass on a per species basis. By considering fisheries as predators, we can use existing ecological indices to measure preference of the fishery for each species. Manly's preference index is:

$$\alpha_i = \frac{l_i/b_i}{\sum_j (l_j/b_j)}$$

where index  $j$  is used for summation over all species (Krebs, 1989). If a species is fished in proportion to its biomass,  $\alpha_i = 1/n$ , where  $n$  is the number of species in the

community. If  $\alpha_i > 1/n$  the species is selected for by fishermen; conversely,  $\alpha_i < 1/n$  means that a species is selected against. The  $\alpha_i$  are normalized to sum to 1, such that preference for a given species is expressed relative to preference for other species in the community. This preference index ( $\alpha_i$ ) was plotted (on a log scale) as bar plot with species names and  $1/n$  as reference level. The same analysis could be repeated for catch (instead of landings) by adding discards to the landings data.

A system-wide metric of the breadth of the fishery was calculated with Levin's measure of niche breadth:

$$\text{Niche breadth} = \frac{1}{\sum_j \alpha_j^2}$$

*Niche breadth* is maximal when all species are fished in proportion to their biomass and decreases when species are increasingly selected for or against (Krebs, 1989). This metric provides a single measure of fishing balance that can be compared among ecosystems.

### 5.2.2 Results

The number of species accounting for 95% of landings biomass varied widely among areas, ranging from nine species in Iceland and the North Sea to 26 species in the Bay of Biscay (Figure 5.3). These numbers reflect both the concentration of the fisheries on particular species as well as the number of available species. Evenness in the catch was corrected for the number of species and therefore reflects the evenness of fisheries across the available species (Table 5.2).

The proportion of each species in the catch depends on its abundance in the sea, value, and fishery regulation, among other factors (Figures 5.3–5.4). In the Bay of Biscay most of the selected species have larger proportions in the landings than in the biomass ( $l > b$ ; Figure 5.5), whereas a few species (e.g. *Scomber scombrus*) have  $l = b$ . According to the selection index  $\alpha$ , the most preferred species are *Solea solea*, *Nephrops norvegicus*, and *Sepia officinalis*, also those with highest market value; by contrast the most underutilized species are *Micromesistius poutassou*, *Trachurus* spp., and *Capros aper*, also the most discarded species in the Bay of Biscay fisheries.

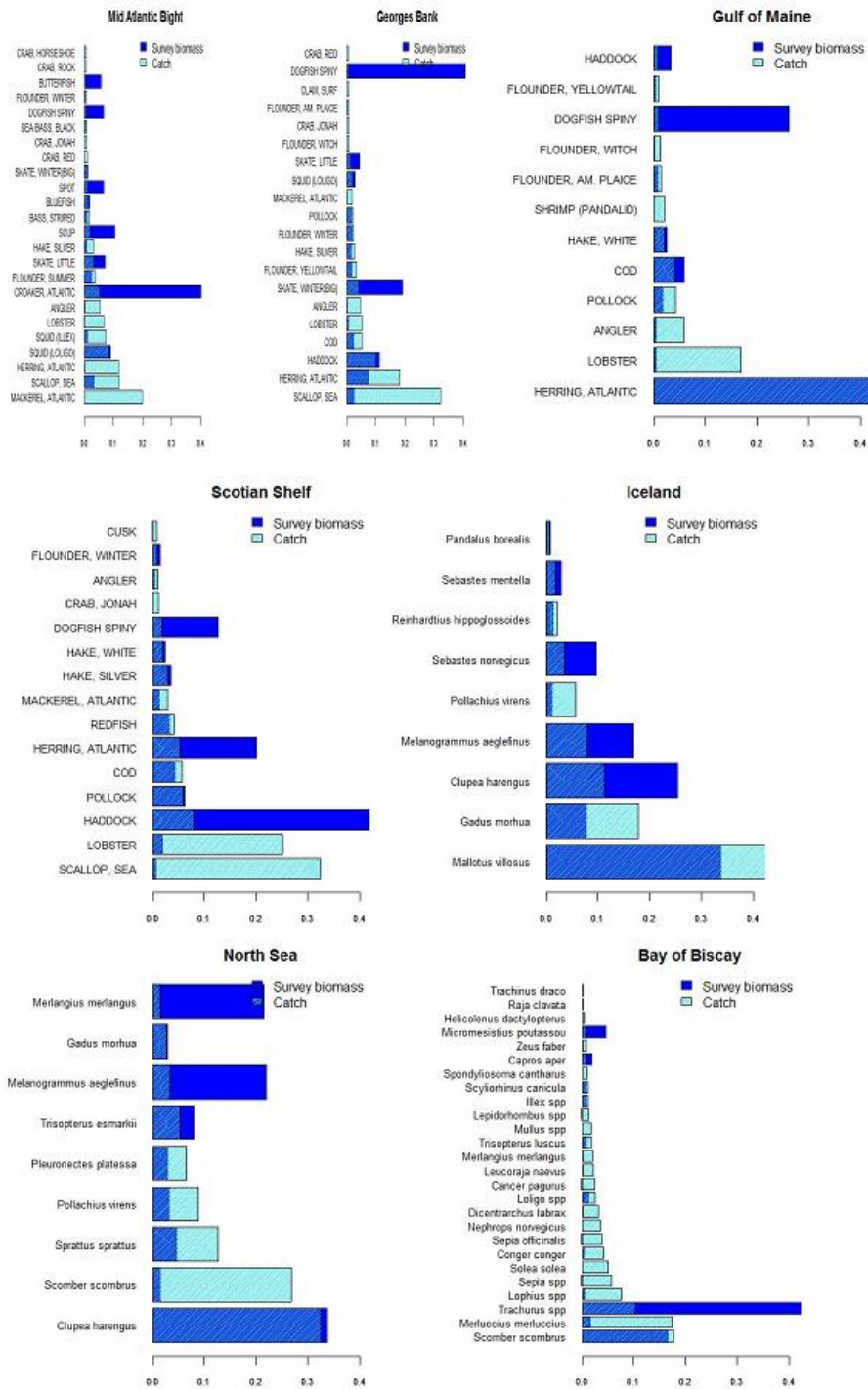


Figure 5.3. Distribution of species biomass in the landings and surveys for each case study. Outliers: Gulf of Maine Herring accounts for 57% of landings and 58% of survey biomass; Iceland capelin accounts for 49% of landings; Bay of Biscay horse mackerel accounts for 68% of survey biomass.

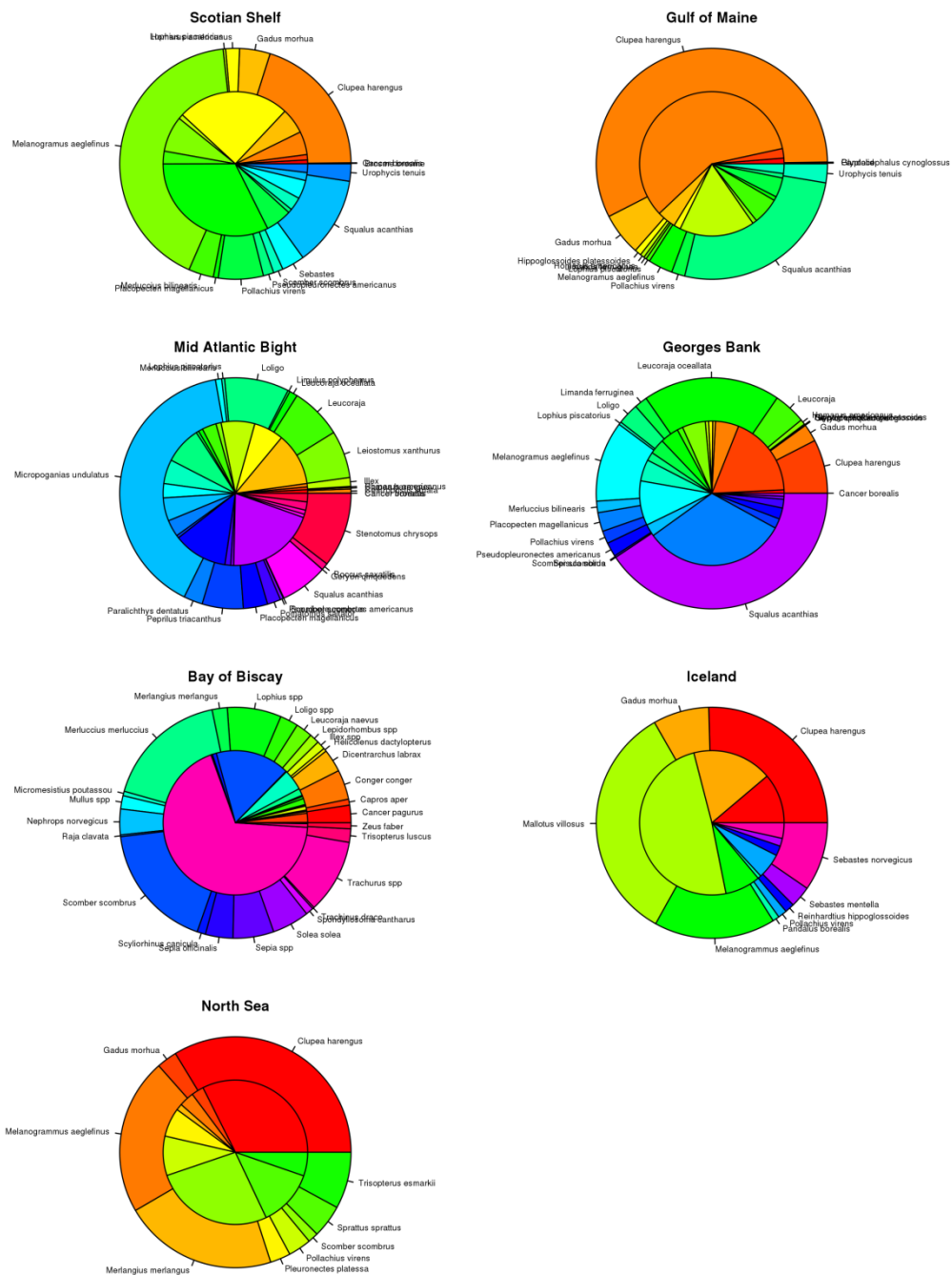


Figure 5.4. Distribution of species biomass in the landings and survey for each ecosystem. The outer pie describes survey biomass species composition, the inner pie landing species composition. Colours are ascribed to one single species within ecosystems (no consistency across ecosystems).

In Iceland the ratios of  $l_i/b_i$  were much closer to unity (Figure 5.5); among this subset of species there is less targeting of preferred species. Relative to their estimated biomass, the most preferred species are saithe (*Pollachius virens*), cod (*Gadus morhua*), and Greenland halibut (*Reinhardtius hippoglossoides*). Least preferred are redfish (*Sebastes norvegicus*), herring (*Clupea harengus*), and haddock (*Melanogrammus aeglefinus*).

In the North Sea the ratios of  $l_i/b_i$  were close to unity for *Pleuronectes platessa*, *Pollachius virens* and *Sprattus sprattus*. *Scomber scombrus* was strongly preferred, while *Merlangius merlangus* and *Melanogrammus aeglefinus* were the least preferred (Figure 5.5).

In New England there is high variability in the ratios of  $l_i/b_i$ , indicating that some species are highly preferred by the fisheries while others are avoided (Figure 5.5). Many of the preferred species are invertebrates: Jonah crabs (*Cancer borealis*), rock crabs (*Cancer irroratus*), red crabs, lobsters (*Homarus americanus*), and sea scallops (*Placopecten magellanicus*), shrimp (*Pandalus borealis*), and surf clams (*Spisula solidissima*). Despite the adjustments for catchability, some of this preference could reflect lack of availability to the survey gear. Preferred finfish species include angler (*Lophius americanus*), witch flounder (*Glyptocephalus cynoglossus*), and mackerel (*Scomber scombrus*). Underutilized finfish species include spiny dogfish (*Squalus acanthius*), herring (*Clupea harengus*) haddock (*Melanogrammus aeglefinus*), cod (*Gadus morhua*), winter skate (*Leucoraja ocellata*), little skate (*Leucoraja erinacea*), butterfish (*Peprilus triacanthus*) and Atlantic croaker (*Micropogonias undulatus*). For the case of cod in the Gulf of Maine, this apparent underutilization is due to regulations to rebuild the depleted cod stock.

Niche breadth ranged from highest in the Bay of Biscay to lowest in the North Sea (Table 5.3). These measures of niche breadth are consistent with and summarize the ratios of  $l_i/b_i$  and the preference index  $\alpha_i$ , across species. Fisheries in the Bay of Biscay and Iceland are much more in balance with the available biomass, compared with the New England regions, where more species are either highly preferred or underutilized (Figure 5.5).

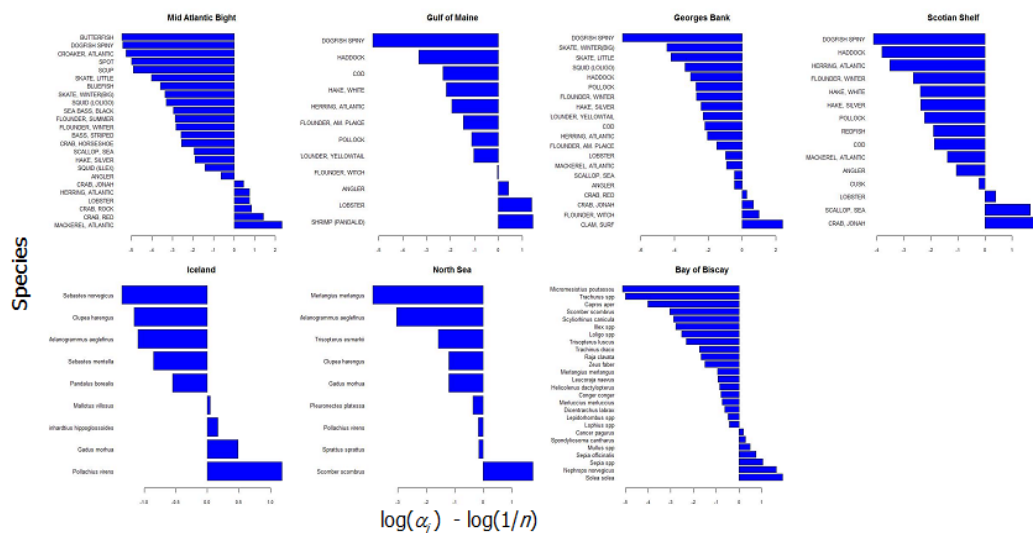


Figure 5.5. Manly's preference indices for major species caught in each ecosystem. Alphas are plotted on a log scale and standardized by  $1/n$ , the theoretical preference index if all species were fished equally. Positive values indicate positive selection.

**Table 5.3.**

AREA	CATCH EVENNESS	NICHE BREADTH
Biscay	0.228	7.23
Iceland		4.92
Mid Atlantic Bight		4.16
Gulf of Maine		3.86
Scotian Shelf		3.30
Georges Bank		2.95
North Sea		2.32

### 5.2.3 Conclusions

WGECO has advanced a methodology to empirically estimate the degree of balance of fisheries in relation to the available biomass. Based on this methodology, the fisheries in Iceland and the Bay of Biscay are more balanced than the fisheries in the North Sea and the four New England regions. The main driver of unbalance in fisheries might be contrast in species value.

These conclusions depend on a number of assumptions. Except for New England, survey biomass estimates were not adjusted for catchability, meaning the selection indices of some species could be influenced by catchability. Even in New England, some species (e.g. surf clams) may not be available to the survey. Results were obviously sensitive to the list of species retained for analyses, which was constrained by data availability. Despite these caveats, the preference indices seem reasonable for most species. Differences in catchability introduce variability of the selection indices but do not drive the main patterns that were observed.

### 5.2.4 Future work

Species that are caught but discarded do not appear in landings data and are therefore underrepresented relative to the total catch biomass. By accounting for the discards of each species, future work could use total catch in place of landings. The use of landings data indicates the degree of balance in marketing and utilization of fish species. The use of total catch data would be more representative of the ecological effects of harvesting on the fish community.

Balanced harvesting also requires species to be fished in proportion to their productivity. If it were possible to estimate the annual production of each species, these calculations could be repeated with production in place of landings.

Here the group estimated the degree of balance as a snapshot in a window of years. The degree of balance in each fishery could be estimated over blocks of years to see if fisheries are becoming more or less balanced.

This approach for examining the distribution of fishing across species could be extended to size distribution with suitable size stratified data in the catch and survey.

Future work could investigate reasons for the degree of balance or lack of balance (e.g. market value and preference, gear restrictions, regulations for rebuilding species, or bycatch). Given this method to estimate the degree of balance in a fishery, does a more balanced fishery impose a smaller impact on the ecosystem? What level of balance is desirable and how does this relate to the overall fishing pressure?



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## **6 Distributional indicators in the context of MSFD: developing operational and surveillance indicators and exploring the link to pressure and drivers (tor d) and request from ICES to determine appropriate methods used to identify significant distribution shifts (ToR h)**

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Understanding past and future distribution of marine fish is necessary to support successful ocean resource management (e.g. Kaiser *et al.*, in press). Under this ToR, WGECO suggests key aspects of distribution which are relevant in the context of biodiversity and/or which are possibly related to pressures, such as fishing, and drivers, such as temperature change. For each of these aspects, possible indicators are listed, and guidance is given on ways to estimate distributional indices from data. Further, surveillance distributional indicators in the context of the Marine Strategy Framework Directive (MSFD) are discussed.

The distribution of marine taxa is highly variable in space and time, as species are affected by biological, environmental and anthropogenic drivers such as the interaction of individual life history and behaviour, recruitment, predation and competition, habitat availability, fishing pressure, seasonality and long-term climate forcing, inter alia (e.g. Cheung *et al.*, 2009; 2013; Simpson *et al.*, 2011; Poloczanska *et al.*, 2013; Englehard *et al.*, 2014; Rutterford, 2015; Hill *et al.*, 2015). Of these drivers, temperature and benthic habitat complexity are important in influencing marine fish distribution (e.g. North Sea cod: Rutterford *et al.*, 2015). Temperature has been demonstrated to have a strong influence on marine fish distribution and, by implication, changes in climate have the potential to exert a significant impact on this distribution, ultimately affecting the ecological and socio-economic dynamics of marine resources (Murawski, 1993; O'Brien *et al.*, 2000; Perry *et al.*, 2005; Brander *et al.*, 2003; 2007). Similarly, for demersal fisheries in particular, benthic habitat complexity is an important component of habitat suitability and can drive the range and distribution of marine taxa (e.g. Diehl, 1992; Kaiser *et al.*, 1999; Grabowski, 2004). Given recent concern over the effects of fishing on fish population size and habitat, growing concern over the potential consequences of climate change on marine environmental conditions, and now the political imperative to monitor change in, and assess the status of, fish distributions in the EU, the need for good models and indicators that describe and quantify changes in fish distribution is immediate.

### **6.1 Key aspects of distribution which are relevant in the context of biodiversity and associated metrics**

Key aspects of distribution were suggested by all members of WGECO following visual inspection of selected distribution maps. The suggestions were grouped into five topics: geographical, occupied area, aggregation, pattern and pattern dynamics. The driver or pressure to which the response is expected to be related will affect the aspect which is of concern as well as the appropriate metric. For example, if the driver investigated is temperature, it seems most relevant to look at the extreme and average temperatures of the observed individuals rather than latitude, as temperature is not always correlated tightly to latitude, in particular when examining smaller regions. If the metric is investigated with the purpose of tracking risk of the species to fisheries induced mortality, aggregation or overlap with fishing activities may be the most relevant metrics. These considerations need to be made in advance of the analy-

sis to ensure that it does not become a 'correlation hunt'. A process to ensure this could be first to consider which aspect of the distribution the driver/metric is likely to affect, then consider which of the possible metrics would be most suited to capturing this change.

In the interpretations of these metrics, it should be kept in mind that different life stages may respond differently to pressures, and it would often be necessary to investigate juvenile and adult distribution separately. Further, in many cases, these metrics seem most relevant at the species level across all geographical regions (e.g. extreme temperatures and latitudes) rather than within subareas.

Distributional changes can take several forms. Link *et al.*, 2011 grouped distributional changes relevant to fish stock management into three main types:

- Parallel shift: The distribution remains of the same extent and shape in space, but is shifted in one direction;
- Contraction/expansion: The distribution is contracting or expanding;
- Splitting/merging: One joint distribution changes into two or more separate distributions by an emerging gap between two higher density peaks or two or more separate distributions change into one joint distribution by closing gaps between existing distributions. Both can happen with or without a change in the extremes of the distribution.

These three main types are all related to the aspects geographical, occupied area, and aggregation which are on the aspects generally investigated in relation to climatic impacts (Johnson *et al.*, 2013). In contrast, most ecological literature investigating biological effects on distribution deals with pattern and pattern dynamics on a much smaller spatial scale (Johnson *et al.*, 2013; Stier *et al.*, 2014).

WGECO has listed potential indicators of each aspect in Table 6.1 along with potential caveats of different indicators, potential uses and which types of distributional changes the metrics reflect. As can be seen from the table, no single indicator will reflect all types of distributional change, and a suite of indicators including metrics of 'geographical extent' and 'aggregation' are needed as a minimum rather than any single indicator alone. There are numerous examples from the literature of studies using average and extreme latitudes and longitude to detect distributional change relating this to climatic changes (Perry *et al.*, 2005; Dulvy *et al.*, 2008; Last *et al.*, 2011; Yemane *et al.*, 2014; Fossheim *et al.*, 2015 are recent examples) while aggregation is typically related to either abundance or risk to exploitation (Rindorf and Lewy, 2012;). Distribution shifts are the most obvious response by fish populations to changes in sea temperature regime (Brander *et al.*, 2003), with warming trends tending to cause northward shifts in range (Rose, 2005). Recent warming of the Northeast Atlantic has coincided with northerly shifts in the distribution of many species generally associated with more southerly latitudes (Quéro *et al.*, 1998; Perry *et al.*, 2005; Beare *et al.*, 2004). However, these shifts in distribution are often only really apparent as changes in distribution when the entire distributional range of the species is mapped.

### 6.1.1 Caveats related to sampling coverage

The key element in developing metrics is whether the survey information available is sufficient to reliably provide the data for that metric. This should not be taken for granted. The distributions of many species, especially when being rare, will be difficult to fully encompass even with many combined surveys, e.g. northern hake. If the surveys do not cover the whole distribution of a species then metrics such as area occupied, distribution centroids or boundary positions will be difficult to determine and be biased against detecting changes on the border or outside the surveyed area. Changes in immigration or emigration may confound these metrics.

There are few species of fish found in surveys that are not found across the NEA. For instance no species caught in the North Sea groundfish surveys is not also found in surveys outside the North Sea. In turn this would suggest that the surveys are generally unlikely to fully encompass the distributions of any one of these species. So, at a species level, many indicators of distribution will be difficult to establish reliably, including ranges, area occupied centroids etc. Many surveys are particularly designed to monitor certain species only, usually commercially important ones, and by this do not allow to quantitatively investigate the distribution area of other species in the survey catches.

Distribution metrics at the subregional or stock level may however be covered and of value to management. For instance, a distribution metric for the cod as a species might be difficult to interpret, but one for the North Sea could be much more valuable. Other cases such as distribution metrics for perch (*Perca fluviatilis*) in the Baltic Sea would also be potentially valuable. While perch are found outside the Baltic, this is the only place where they are found widely distributed across the coastal area. However, perch distribution within the Baltic Sea might be difficult to interpret since it is likely to be driven by both environmental effects like temperature and anthropogenic causes like eutrophication or fishing. Distribution metrics would especially be valuable for non-commercial and particularly for vulnerable species, as required under MSFD Descriptor 1. In particular, this would be important where the distributional metrics showed changes in relation to distribution of fishing effort of the relevant métier, derived from VMS data. So a change in a species distribution could make it more or less vulnerable to incidental capture in commercial fisheries depending on the degree of spatial overlap.

In all these cases the role of the indicators would be as surveillance indicators, and the first action when changes were seen in the metric would be to examine the maps, rather than take action on the basis of the metric. Operational decisions could then be based on the mapped distribution.

### 6.1.2 Estimating distributional indices from data

Several of the metrics in Table 6.1 are affected by the methods of modelling species distribution used or whether raw data are used rather than smoothed distributions. A few of the metrics require raw data, but where this is not the case, it may be preferable to base the indices on smoothed distributions to limit the effect of between haul variation in catches which is not related to large-scale distribution. In this section we review methods to describe changes in the distributions of key fish species to establish the extent to which biological, environmental or anthropogenic influences have been the main drivers of change using species distribution modelling.

Section 3 of this report provides lessons learned for predictive modelling. Species distribution models (SDM) can estimate the relationship between species records at

different locations and the environmental and/or spatial characteristics of those sites (Franklin, 2009). Species distribution models are useful for extending typically sparse point observations to create continuous predictions of species' distribution or habitat type. Species distribution models have been used extensively in both terrestrial and marine environments to make contemporary distribution maps (e.g. Dolan *et al.*, 2008; Guinan *et al.*, 2009; Tittensor *et al.*, 2009; Yesson *et al.*, 2012; Knudby *et al.*, 2013 a,b,c; Rooper *et al.*, 2014). They have been used to predict species/habitat responses to climate change (e.g. Lawler *et al.*, 2009) and to predict the future range of invasive species (Peterson and Robins, 2003; Peterson, 2003). They can also provide knowledge of the differences between actual and potential species distribution, making them a very useful tool for habitat restoration in fisheries and ocean management. There is a large range of Species distribution models and modelling approaches are currently available (Guisan and Zimmerman, 2000). More recently, machine learning methods have become more common. These methods comprise of a series of non-parametric techniques capable of synthesizing regression or classification functions based on the available data. Random Forest (RF) (Breiman, 2001) is one such method that can be used for regression or classifications and is considered superior to most commonly used methods (Cutler *et al.*, 2007). Others have used regression based techniques such as generalized additive (GAM) modelling (e.g. Rooper *et al.*, 2014) which offer a different theoretical basis for the production of the interpolated surfaces. It is also important to establish the lower data requirement limits that can support reliable species' mapping and generation of distribution indicators. Rare/threatened and declining species are often the subject of conservation concern and management measures. Capacity to assess changes in the distributions of such species is critical if human impacts, or the efficacy of mitigation measures, are to be assessed adequately. By definition such species are generally scarce and sample data consist primarily of zero/low abundance observations. Such data are difficult to fit to spatial models. It is important to establish the data limitations that permit reliable mapping and hence identify those species for which reliable distribution indicators can be determined and those species that are simply too scarce to support distributional assessment. If distribution indicators are to be used to support implementation of the MSFD, it will be necessary to provide a clear statement regarding which species can, and which species cannot, be assessed and evaluated reliably. The maps produced should be subjected to cross validation techniques to assess the reliability of the mapping process. Pooling annual datasets to improve data reliability and mapping potential could be an option. Another approach would be to employ a species archetypes model (SAM), and group species into "archetypes" according to their environmental responses using a finite number of regression models, this can be done for binary data or count/biomass data (Dunstan *et al.*, 2013).

In general, temporal patterns derived from smoothed surfaces made from different spatial models tend not to differ widely where clear changes occur, the survey coverage is substantial and the error distribution of the observations is appropriately modelled (compare for example Lewy and Kristensen, 2009 with Rindorf and Lewy, 2012). In areas with limited data and outside the sampled area, the smoothed surface tend to differ more widely and in general be less reliable, making the choice of model is one of the determining factors in the resulting conclusions. This should be avoided whenever possible, for example by not extrapolating beyond the surveyed area.

Table 6.1. Aspects important to analysis of distribution, possible metrics, caveats and examples of drivers and pressures to which the metric may respond.

HIGH LEVEL ASPECT	DETAILED LEVEL ASPECT	POSSIBLE METRICS	CAVEATS	RESPONSIVE TO WHICH DRIVERS AND PRESSURES
Geographical extent	Latitude and longitude of distribution area Depth in distribution area Temperature in distribution area	Extreme, 5% and 95%tiles, and average of latitude, longitude, depth and temperature of recorded specimens	Highly dependent on surveyed area and hence not comparable if surveyed area has changed  The accuracy of the estimated distribution relies on catchability being independent of place, depth and temperature	Responsiveness to climate change is likely to be greatest for temperature based metrics and metrics which are highly correlated to the limiting factor. Responsive to spatially targeted fishing pressure in species which are not highly mobile such as pelagics, the most appropriate metric depending on how the fishing pressure is aggregated (by latitude, longitude, depth or temperature). All metrics may be responsive to changes in total abundance as well as to climate change and fishing.  Geographical extent can change without a concurrent change in other aspects of distribution.  Can be used to detect: Parallel shift and Contraction/expansion
Occupied area	Potential and realised habitat	Surface area of empty areas (potential and realised) Surface area of near-empty areas (5% lowest abundance) (potential and realised) Surface area of occupied areas (potential and realised) Realised occupied area relative to potential occupied area	The occupied area when measured directly from the number of empty samples recorded is highly statistically dependent on abundance. If this effect is not desired, modelled probability of observing the species can be used together with a threshold defining empty areas (e.g. probability of observing the species is less than 10%)	Responsive to habitat loss (including that induced by climate change) and a general decline in habitat suitability which may lead to poorer habitat being vacated. Can be responsive to spatially targeted fishing pressure in species which are not highly mobile. All metrics may be responsive to changes in total abundance.  Occupied area can change without a concurrent change in geographical distribution. However, most aggregation measures depend on the amount of empty habitat and hence are related to occupied area.  Can be used to detect: Contraction/expansion

HIGH LEVEL ASPECT	DETAILED LEVEL ASPECT	POSSIBLE METRICS	CAVEATS	RESPONSIVE TO WHICH DRIVERS AND PRESSURES
Aggregation	Surface area of high and low density areas	Area containing a fixed percentage of the population (both high and low) standard deviation of average latitude, longitude, depth and temperature	Area containing a fixed percentage of the population is statistically dependent on abundance unless data are model smoothed before estimation.	Can be responsive to spatially targeted fishing pressure in species which are not highly mobile. All metrics may be responsive to changes in total abundance. Indicative of stock sensitivity to potential changes in overlap with the fishery as a highly aggregated species will experience a large increase in pressure if aggregation areas become targeted by the fishery. Aggregation can change without a concurrent change in geographic aspects of distribution. However, most aggregation measures depend on the amount of empty habitat and hence are related to occupied area. Can be used to detect: Contraction/expansion and Splitting/Merging
	Patchiness	Lloyds patchiness index/Negative binomial k	Patchiness indices require a substantial number of non-zero observations to be reliable	As above

HIGH LEVEL ASPECT	DETAILED LEVEL ASPECT	POSSIBLE METRICS	CAVEATS	RESPONSIVE TO WHICH DRIVERS AND PRESSURES
Pattern description	<p>Number of patches</p> <p>Distance between patches</p> <p>Size/area of individual patches</p> <p>Density in patches</p> <p>Variance in density on different spatial scales/Evenness of patches</p>	<p>Number of patches</p> <p>Number of isolated patches</p> <p>Distance between patches</p> <p>Size/area of individual patches</p> <p>Density in patches</p> <p>Variance in density on different spatial scales/Evenness of patches</p>	<p>Patch definition is still very much a subjective decision and spatial extent and distance between observed patches is highly dependent on sampling distribution relative to patch size. The spatial scale of sampling is crucial to the determination of patch size, as the minimum size is determined by the minimum distance between sampling locations.</p>	<p>Patch number, size and density depend on patch 'birth' and 'dead' rate. Distance between patches and number of isolated patches are both related to number of patches, whereas variance in density in patches at larger spatial scales is related to aggregation.</p> <p>Pattern can potentially change without a concurrent change in geographic aspects of distribution. Aggregation and occupied areas are related to pattern if measured at the same spatial scale. However, the spatial scale of patterns is often less than that investigated for aggregation and occupied area.</p> <p>Can be used to detect: not simply related to any of the types Parallel shift, Contraction/expansion or Splitting/Merging</p>
Pattern dynamic	<p>Within and between patch dynamics</p>	<p>Connectivity/contagion between patches/areas</p> <p>Increase or decrease of individual patches</p>	<p>Patch definition is still very much a subjective decision and spatial extent and distance between observed patches is highly dependent on sampling distribution relative to patch size. The temporal scale of sampling determines which temporal scale of dynamics can be investigated.</p>	<p>Connectivity is related to species mobility and distance between patches. Increase and decrease of individual patches is related to patch density and size/area.</p> <p>Can be used to detect: not simply related to any of the types Parallel shift, Contraction/expansion or Splitting/Merging</p>



## 6.2 Surveillance indicators for the MSFD

The MSFD requires EU Member States to achieve Good Environmental Status (GES) for 11 Descriptors (D) of GES by 2020. Under descriptor 1, two key indicators focus on aspects of species' distribution: "Distributional range" and "Distributional pattern within the latter". To assess the potential of different distributional indicators to meet MSFD monitoring and assessment needs, distribution range and pattern indicators have been applied to fish survey data, concluding that neither responded in a consistent way to a release in fishing pressure and that substantial unexplained variation remained in both when applied to sensitive, less abundant species (Greenstreet *et al.*, 2012). A second study examined numerous indicators to explore abundance-distribution relationships, and concluded that few were unbiased when based directly on observations rather than smoothed values and some produced erroneous relationships when related to abundance (Rindorf and Lewy, 2012).

The range of potential metrics available to fulfil the MSFD distribution indicator roles is extensive, but in many cases exactly what these metrics convey is not clear making proper interpretation difficult. Often a singular specific change in a distribution metric can arise as a result of several different types of change in the actual distribution. Conversely, the actual distribution might change in different ways, some of which have no effect on the distribution metric if only a single metric is used. Thus a key means of addressing these problems is to use a suite of distribution metrics so that several possible types of change can be identified. For example if the species became more aggregated, this may not change a metric related to geographical extent but would change a metric related to aggregation.

None of the distribution metrics listed here convey a minds-eye image of what a map would look like. Hence, the metrics are seen as a tool to derive surveillance measures to alert users/managers to investigate the actual distribution on maps. The metrics listed in Table 6.1 cannot in themselves form the basis of advice regarding management measures and hence are not operational indicators in the sense of the MSFD. They are however considered to be useful surveillance indicators: If a metric, or probably a number of metrics change at the same time, or are trending, this should ring an alarm bell triggering a more detailed investigation of the type of change in distribution and the causes of this. The identification of drivers of change is crucial to managers to respond to it appropriately. The suite of indicators should be chosen such that if changes in actual distribution were to occur, that at least one surveillance indicator in the suite would respond to this change in such a way as to trigger a "management reaction".

The surveillance indicator approach can be used to monitoring potential distribution change where the production and interpretation of all the maps involved would be too arduous or resource demanding to undertake on a routine basis. In this case, if a management reaction is triggered by change in one or more of the surveillance indicators, in the first instance, this reaction should involve the inspection of the relevant maps so that a clear appreciation of the actual distribution change that has taken place can be established. Only distribution maps convey all the information that management would need in order to: a) establish that real problems have occurred, and b) inform as to appropriate management measures.

The MSFD requires assessment of GES at the subregional scale, e.g. the Greater North Sea, or at the regional scale, e.g. the Northeast Atlantic. In most cases indicators will be reported at the subregional scale and this is appropriate where these indicators

relate to communities, which generally vary at subregional or smaller scale. However, many fish populations cover areas that exceed the spatial extent of MSFD-defined subregions. For example Cuckoo ray can be found from Gibraltar to the north coast of Norway; their range therefore covers at least three MSFD-defined subregions and survey data have been collected from three continental shelf subregions within the NE Atlantic Region: The Greater North Sea; The Celtic Seas; and the Bay of Biscay and Iberian Coast. This begs the question what is the appropriate scale on which to describe the distributions of fish.

Over the last century the NE Atlantic has experienced several fluctuations in water temperature and salinity. Fishing pressure in the region steadily increased through to the mid-1980s, then subsequently decreased (Greenstreet *et al.*, 2011). These environmental and anthropogenic drivers have either directly influenced the distribution of many fish species or had an indirect effect through impacts on species' population abundance and/or on their habitat. Having derived scientifically robust indicators and established the appropriate spatial scale on which distribution should be assessed these indicators can now be applied to historic survey data to investigate the relative importance of natural processes and anthropogenic activities as drivers of change in the distribution of key fish species.

### 6.3 ICES Special Request

WGEKO was requested to advise on possible methods to detect distributional change for a range of species in the Northeast Atlantic (Table 6.2) and the group identified species and survey combinations where metrics of geographical extent, occupancy and aggregation could reliably be evaluated on a species and stock level. For instance in western waters it should be possible to evaluate movement in the southern limit for some northern species such as haddock or cod. Northern limits for other species e.g. sardine should also be possible to determine. However, for other species, it will be necessary to limit the analysis to subregional distribution due to the lack of overlap in time and space between different surveys.

For each species and, where relevant, subregion, WGEKO recommends including metrics of each of the three aspects geographical extent, occupied area and aggregation to detect large-scale distribution shifts, including parallel shifts, expansion/contraction and splitting/merging. The metrics should preferably be based on smoothed distributions assuming an appropriate error distribution (e.g. non-symmetrical such as delta or negative binomial). An appropriate selection of indicators to quickly detect geographical trends in distribution over time could be 5% and 95% tiles, and average of latitude, longitude, depth and temperature of recorded specimens (Geographical extent), surface area of realised occupied areas (Occupied area), the area containing 95% of the population and standard deviation of average latitude, longitude, depth and temperature (Aggregation). As these metrics are likely to be highly correlated, they should be combined in a multivariate analysis of the trend or in a composite indicator using e.g. principal component analyses prior to determining the trend to avoid detecting spurious correlations.

WGEKO recommends that care is taken when using the survey data available in DATRAS. Combining the surveys on a regional level without accounting for the differences in each survey could lead to a misinterpretation of the information they provide. Surveys shouldn't be combined without first addressing issues of catchabilities between surveys and standardizing the survey areas both temporally and spatially. The surveys vary spatially and temporally, survey goals and objectives have changed

over the years and the fishing gear has changed on many vessels. For example in the Baltic Sea although the Baltic International Trawl Survey series starts in 1991, there was a gear change in 2001. In the Celtic Seas there are a number of surveys which vary temporally, the West Coast of Scotland has data as far back as 1985, but the most recent groundgear change occurred in 2009. The current Irish Groundfish survey began in 2003, with but spatial coverage changed in 2005 (it moved out of the Irish Sea and increased its stations on the shelf). The Northern Ireland Groundfish survey is available in DATRAS from 2008 onwards, but the Northern part of the Irish Sea and St Gorges Channel are surveyed using two separate survey designs and they are not directly comparable. In the North Sea data are available in DATRAS from 1965 for the first quarter survey and 1987 for the third quarter survey. There has been several changes in survey design in the history of this survey. It is perhaps more sensible to use first quarter data from 1983 and third quarter data from 1998, (although it is considered possible to use third quarter data from 1991, with less confidence attached). For the Bay of Biscay, the EVHOE survey has been standardized from 1987 onwards, but moved into the Celtic Sea at a later date. For the Iberian coast, the Portuguese data should only be used from 2005, but it should be noted that they are currently awaiting a new vessel and their gear type is likely to change with the new vessel to be more comparable to adjacent surveys. The Spanish data submitted to DATRAS contain only a standard species list, and it would be unwise to use these data to infer a southern range for species not "present" in these data as this may not be correct given that all of the species are not reported. Each of these surveys use different gear types and most are not directly comparable to each other. There are similar issues with the Beam Trawl surveys, the "BTS" data contains information on five national surveys, three different surveys from England and one from the Netherlands and one from Germany. Each of these surveys have a unique gear type and different temporal and spatial scales, so should not be combined without comparisons and calibrations. Care should be taken on using any of these surveys without prior quality control on the DATRAS data such as that in Moriarty and Greenstreet (in preparation).

**Table 6.2. Identified species and survey combinations where metrics of geographical extent, occupancy and aggregation could reliably be evaluated on a species and stock level, focusing on the Northeast Atlantic.**

SPECIES	APPROPRIATE SURVEYS
Anchovy	Acoustic surveys and bottom-trawl surveys
Anglerfish	Dedicated angler surveys, Bottom-trawl surveys can be used but are weak.
Blue whiting	Acoustic (PGIPS), juveniles from bottom-trawl surveys
Cod	Bottom-trawl surveys
Common sole	Beam trawl surveys. Bottom-trawl surveys can be used but are weak.
Greenland Halibut	Dedicated halibut surveys
Haddock	Bottom-trawl surveys
Hake	Bottom-trawl surveys
Herring	Acoustic surveys, juveniles from bottom-trawl surveys
Horse Mackerel	Juveniles from bottom-trawl surveys, egg surveys for adults
Mackerel	Juveniles from bottom-trawl surveys, egg surveys for adults
Megrim	Dedicated angler surveys, Bottom-trawl surveys can be used but are weak.
Norway pout	Bottom-trawl surveys
Plaice	Beam trawl surveys. Bottom-trawl surveys can be used but are weak.
Pollack	Bottom-trawl surveys
Saithe	Bottom-trawl surveys for adults
Sprat	Acoustic surveys, bottom-trawl surveys can be used but are weak in some areas
Spurdog	Bottom-trawl surveys are weak due to occasional large catches
Whiting	Bottom-trawl surveys

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## 7 Review and further develop indicators of state of sensitive species throughout the ICES area and suggest risk based advice on sensitive species (ToR e)

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Species' life-history traits provide a good indication of their capacity to cope with additional mortality, and can therefore be used to assess their sensitivity to human activities that raise mortality rates above those normally associated with natural ambient environmental conditions. For example, species with "slow-type" life-history traits (large-bodied, slow growing, late age and large size at first maturity, low fecundity, etc.) are particularly sensitive to the additional mortality associated with fishing activity (Jennings *et al.*, 1998; Gislason *et al.*, 2008; Hobday *et al.*, 2011; Le Quesne and Jennings, 2012). Many elasmobranch species in particular are characterised by "slow-type" traits and populations of many elasmobranchs in the North Sea had declined markedly by the 1970s (Frisk *et al.*, 2001; Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999; Walker and Hislop, 1998; van Strien *et al.*, 2009). Teleost species with similar life histories also declined (Philippart, 1998; Rijnsdorp, *et al.*, 1996). Life-history trait composition among the demersal assemblage as a whole had become "faster" by the 1960s (Jennings *et al.*, 1999; Greenstreet and Rogers, 2000; 2006; Greenstreet *et al.*, 2012a).

The method developed by Greenstreet *et al.* (2012) related explicitly to the North Sea demersal fish assemblage sampled by the First Quarter (Q1) international bottom-trawl survey (IBTS). This is an otter trawl survey and as such is likely to have low sampling efficiency for many small demersal and flatfish species that strongly associate with the seabed (Fraser and Greenstreet, 2007). The inclusion of data from three North Sea beam trawl surveys, as well as data from the Q3 IBTS and French Channel groundfish surveys, which use the GOV otter trawl, should therefore extend the previous North Sea demersal species inventory. In addition, pelagic fish species sampled in all six surveys are also now considered. The addition of species sampled from a further thirteen surveys, using both beam and otter trawl, carried out in the Celtic Seas, Bay of Biscay and Iberian Coast, and the wider Northeast Atlantic are also included, giving a total species inventory of 469 species determined from nineteen separate groundfish survey carried out across the continental shelf and shelf edge waters of the Northeast Atlantic, the from Bay of Cadiz in southern Spain to the northeast edge of the North Sea, including the English Channel and Kattegat and Skagerrak. Extending the area of coverage in this way was a primary objective of the work conducted in WGEKO in 2016, to demonstrate that the method for defining suites of sensitive species used by Greenstreet *et al.* (2012) is applicable across the whole OSPAR area of the Northeast Atlantic.

However, two potential shortcomings in the Greenstreet *et al.* (2012) approach need first to be addressed. The first issue concerns the availability of the life-history data required to support the approach, and the methods used to estimate parameter values where these were missing. Instances where all four life-history trait parameters are available are in the minority; of the 469 species considered here, data for all four life-history traits were only available for 140 (30%). Conversely, in 174 instances (37%) all four parameter values were absent, leaving 155 instances (33%) where between one and three of the parameter values were missing.

Greenstreet *et al.* (2012) examined the interrelationships between the four parameters, and between both ultimate body length and length at maturity parameters and the

maximum recorded length of each species, in order to establish a protocol for estimating parameter values where these were missing (see also ICES (2015) for further details). But in using these interrelationships to estimate missing parameter values, only simple models were used. For species where  $L_{\infty}$  and  $K$  were not known, the  $L_{\infty}$  parameter was estimated as a function of  $L_{MAX}$  using a linear regression model derived from data for species where values for both variables were available. Since, the two von Bertalanffy variables  $L_{\infty}$ , and  $K$ s were known to be correlated (Gislason *et al.*, 2008), a second regression model was developed to estimate  $K$  from the estimated  $L_{\infty}$ . The parameters derived from this second model fell well within the 95% confidence limits given for the equivalent relationship by Gislason *et al.*, (2008). Gislason *et al.* (2008) also estimated length-at-first-maturity ( $L_{mat}$ ) as a function of von Bertalanffy ultimate body length ( $L_{\infty}$ ). Greenstreet *et al.* (2012) therefore also developed linear regression models to estimate length-at-first-maturity ( $L_{mat}$ ) for species where this information was missing using both maximum recorded length ( $L_{MAX}$ ) and ultimate body length ( $L_{\infty}$ ) as the explanatory variable; the ( $L_{\infty}$ ) model was deemed the more reliable and therefore used to estimate  $L_{mat}$  where this information was missing. Where age-at-maturity ( $A_{mat}$ ) data were missing, these were estimated by substituting the three other parameters into the von Bertalanffy growth equation rearranged to solve for  $t$ ,  $A_{mat}$ , when  $L = L_{mat}$ . Since  $L_{MAX}$  data were available for all 119 North Sea demersal species considered, estimates for all four life-history parameters could ultimately be determined following the pathway of steps illustrated in Figure 7.11. Given the use of only a single simple model for each estimation process, it is clear that solutions for many species will be identical; where all life-history parameters are missing, species with a given  $L_{MAX}$  will have identical sets of values for the four parameters.

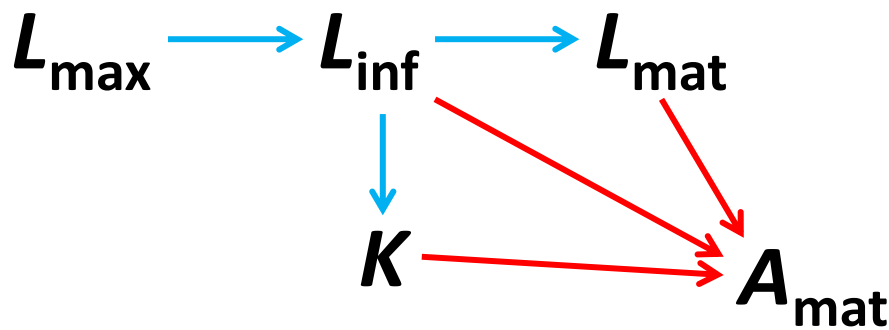


Figure 7.1. Parameter estimation pathway used by Greenstreet *et al.* (2012). Blue arrows represent linear regression models, red arrows represent the use of all three parameters in the von Bertalanffy growth equation rearranged to determine age at maturity from length-at-maturity.

Second, having established, and then made use of the relationships between the four parameters, the species sensitivity metric was the arithmetic mean of the four parameter values. In weighting each parameter equally, and calculating the average across the four parameters for each species, the implicit underlying assumption is that each parameter was independent of the others; this demonstrably was not the case.

In this tor therefore, the question as to how suites of ‘sensitive’ fish species should be defined to support the MSFD D1 Abundance and Biomass indicators for the fish community ecosystem component across the OSPAR area is revisited. The problems involved in estimating the required life-history parameter values where the necessary

information is not available are re-examined. A more robust modelling process is developed, with a key aim of explaining more of the residual variation unaccounted for by the simple models used by Greenstreet *et al.* (2012). In this way a suite of different models is developed that provides many more unique solutions to parameter estimation. Further, an alternative species sensitivity metric is developed that takes account of the interrelationships between the four life-history parameters. Finally, given the fact that ultimate body size is a critical factor in determining species sensitivity to additional mortality, and given that maximum recorded body length is invariably present, we examine whether simply using  $L_{MAX}$  data as the sensitivity metric might not provide an alternative option.

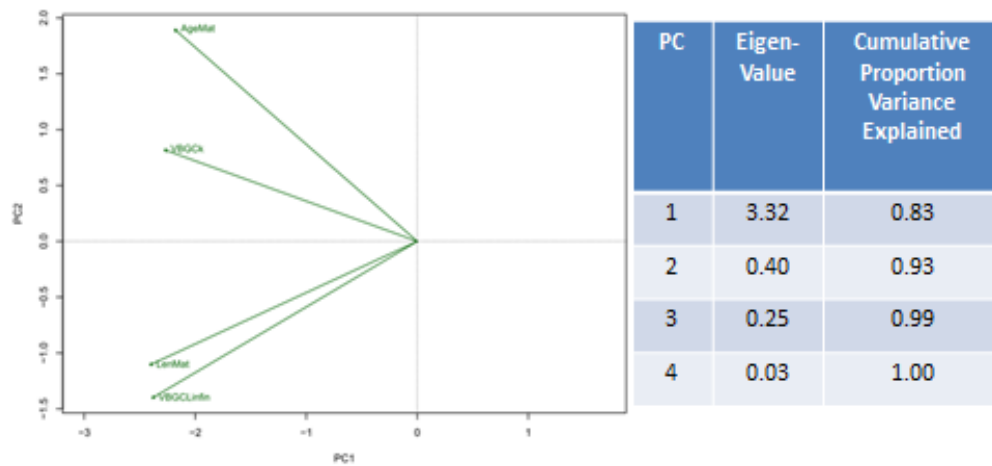
### 7.1 Estimating missing life-history parameter values

To derive ‘sensitivity’ index values for all 469 fish species recorded in the nineteen groundfish surveys carried out in the Northeast Atlantic, information on the two von Bertalanffy growth equation parameters, ultimate body length ( $L_{\infty,s}$ ) and the growth parameter ( $K_s$ ), and length ( $L_{mat,s}$ ) and age at first-maturity ( $A_{mat,s}$ ) were compiled for each species ( $s$ ) from a number of sources (e.g. Jennings *et al.*, 1998; Jennings *et al.*, 1999; Gislason *et al.*, 2008) and the FishBase website ([www.fishbase.org](http://www.fishbase.org)). Table 7.1 illustrates the extent of data availability for the whole species inventory, and for the species inventories likely to be found in each marine region.

$L_{inf}$	$K$	$L_{mat}$	$A_{mat}$	$N$	N. Atl.	N.S.	C.S.	BBIC
😊	😊	😊	😊	140	81	112	124	126
😊	😊	😊	❌	45	12	19	32	43
😊	😊	❌	😊	16	6	12	13	16
😊	😊	❌	❌	46	16	20	29	42
❌	❌	😊	😊	7	3	4	5	6
❌	❌	😊	❌	36	19	17	28	34
❌	❌	❌	😊	5	2	4	4	4
❌	❌	❌	❌	174	67	71	115	158
<b>Number of species</b>					<b>206</b>	<b>259</b>	<b>350</b>	<b>429</b>
<b>% full data</b>					<b>32.5%</b>	<b>27.4%</b>	<b>32.9%</b>	<b>36.8%</b>
<b>% no data</b>					<b>39.3%</b>	<b>43.2%</b>	<b>35.4%</b>	<b>30.8%</b>

Table 7.1. Availability of life-history parameter data for the full inventory of 469 species recorded in nineteen groundfish surveys operating across the Northeast Atlantic, and for the likely species inventories in the Deep North Atlantic (N. Atl.), Greater North Sea (N.S.), Celtic Seas (C.S) and Bay of Biscay and Iberian Coast (BBIC) areas that generally correspond to the MSFD subregions and OSPAR regions II, III, IV, and V. The likely species inventory for each subregion, and the percentages with full life-history parameter information and no life-history parameter information are shown.

Principle Components Analysis confirmed that, not only were all four life-history parameters closely correlated, but that there was also distinct clustering consisting of close correlations between  $L_{\infty}$  and  $L_{mat}$ , and between  $K$  and  $A_{MAT}$  (Figure 7.2). Given the close correspondence between  $L_{\infty}$  and  $L_{mat}$ , this relationship was examined first (Figure 7.3). The relationships for both the transformed and non-transformed data are shown. Since analysis of the residuals was essential to the development of improved parameter estimation models, normalising and homogenising the residual variation was critical. In the non-transformed data (left panel), it is clear that variance in  $L_{mat}$  increases steadily as  $L_{\infty}$  increases, but that this was controlled by log-transformation (right panel) Analyses therefore progressed using the log-transformed data. General Linear Modelling suggested a further significant phylogenetic effect, inferring that the  $L_{\infty}$  -  $L_{mat}$  relationship for Elasmobranchii species differed from the relationship for Actinopteri, Holocephali and Petromyzonti species combined (Figure 7.4), facilitating the use of two separate models to estimate  $L_{mat}$  from  $L_{\infty}$  (Figure 7.4).



**Correlation matrix**

	Linf	k	LenMat	AgeMat
Linf	1	0.78112866	0.95940365	0.6639029
k	<b>0.78112866</b>	1	0.73924621	0.75070572
LenMat	<b>0.95940365</b>	0.73924621	1	0.73857981
AgeMat	0.6639029	<b>0.75070572</b>	0.73857981	1

Figure 7.2. Results of Principle Components Analysis examining the interrelationships of the four life-history parameters.

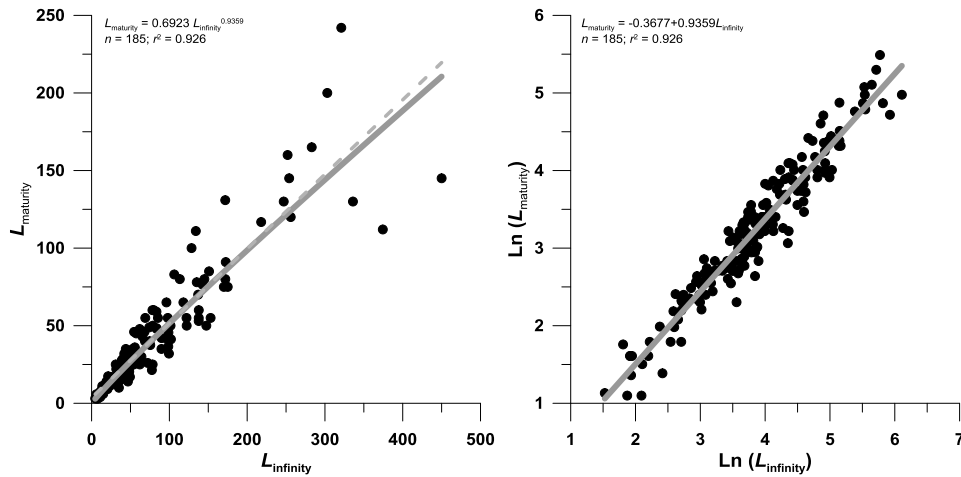


Figure 7.3. The relationship between  $L_{\infty}$  and  $L_{mat}$ . Left panel shows the untransformed data and the power function equivalent to the linear relationship fitted to the natural log-transformed data in the right panel.

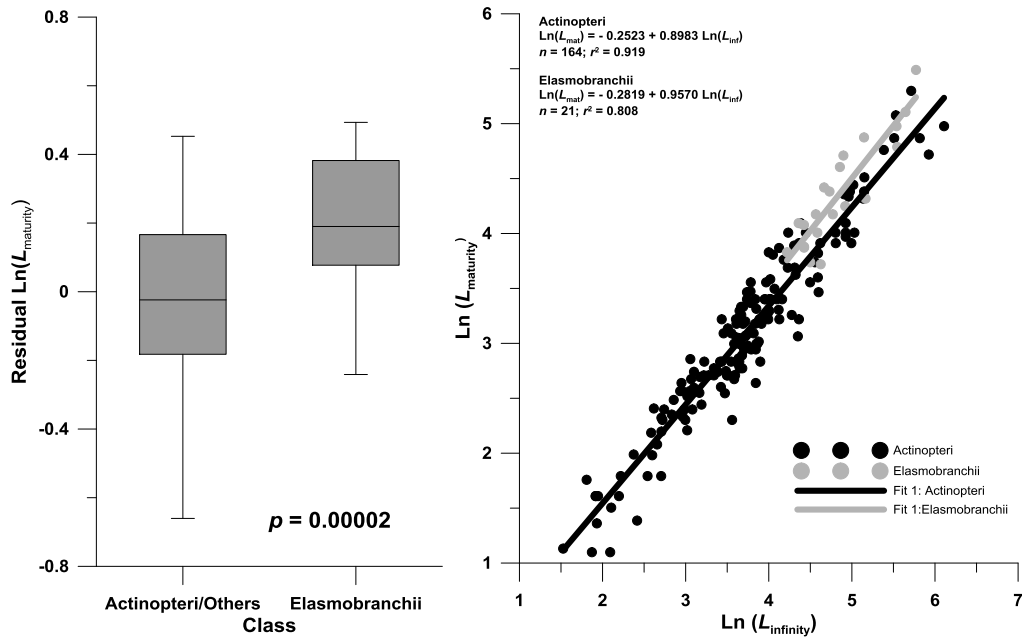


Figure 7.4. Left Panel: Box and Whisker plots showing distributions of the residual variation in the  $L_{\infty} - L_{mat}$  relationship for Actinopteri and the other classes combined and for Elasmobranchii. Right Panel: Resulting separate models for estimating  $L_{mat}$  from  $L_{\infty}$  for Actinopteri and the other classes combined and for Elasmobranchii.

Next the relationship between  $L_{\infty}$  and  $K$  was examined and although highly significant, considerable residual variation was left unexplained (Figure 7.5). Again to control the variance heterogeneity, log-transformed data were analysed. Analysis of residuals suggested two significant effects: from  $A_{mat}$  and from Ecotype. However,  $A_{mat}$  had the stronger influence and when included in the GLM, the significance of the Ecotype effect was eliminated. The strength of the effect of  $A_{mat}$  was expected given the close correlation between  $A_{mat}$  and  $K$  identified in the PCA (Figure 7.2). The strong relationship between  $A_{mat}$  and  $K$  can be explained on theoretical grounds given the tight relationship between  $L_{\infty}$  and  $L_{mat}$  illustrated in Figures 3 and 4, which infers that  $L_{mat}$  is a fairly restricted fraction of  $L_{\infty}$ . Figure 7.6 illustrates this, showing von Ber-

tanaffy growth curves for two species with identical  $L_\infty$  values but with values of  $K$  differing by a factor of two. In both cases  $L_{mat}$  is assumed to be the same fraction of  $L_\infty$ , and therefore identical. Where this is the case,  $A_{mat}$  and  $K$  are directly inversely proportional; as  $K$  doubles,  $L_{mat}$  halves. A fixed effects General Linear Model of the form,

$$\text{Ln}(K) = \text{Ln}(L_\infty) + \text{Ln}(A_{mat}) + \text{Ln}(L_\infty):\text{Ln}(A_{mat}) \text{ (AIC 174)},$$

could therefore be used when  $A_{mat}$  data were available and estimates  $L_\infty$  had been derived. Comparison of model derived estimates of  $K$  with actual recorded values suggested that the model performed reasonably well, but that three of the four extreme high outliers of  $K$  had not been captured (Figure 7.7). When  $A_{mat}$  data were not available, the alternative GLM including the factor 'Ecotype' could be used. Analysis of the residuals in Figure 7.5 (right panel) suggested a significant difference in the relationships between  $L_\infty$  and  $K$  between the Bathydemersal and Bathypelagic groups combined and each of the Benthopelagic, Pelagic and Demersal ecotypes (Figure 7.8: left panel), facilitating the use of four separate models to estimate  $K$  from  $L_\infty$  (Figure 7.8: right panel).

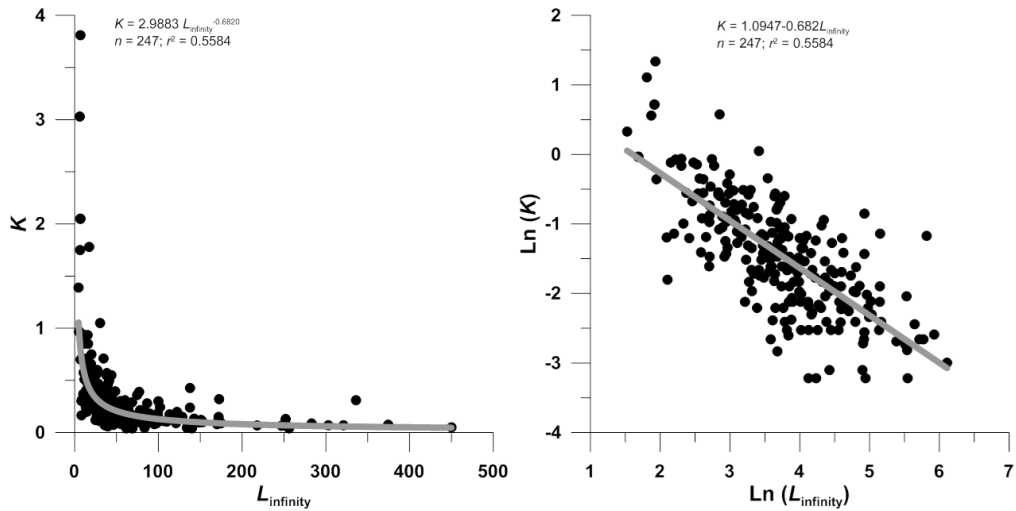


Figure 7.5. The relationship between  $L_\infty$  and  $K$ . Left panel shows the untransformed data and the power function equivalent to the linear relationship fitted to the natural log-transformed data in the right panel.

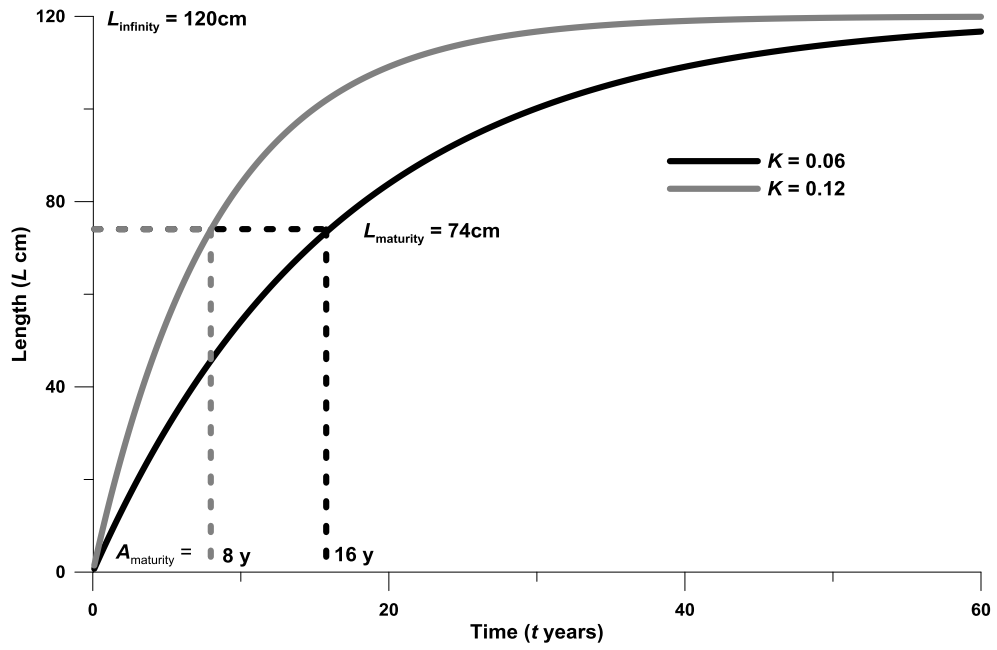


Figure 7.6. Von Bertalanffy growth curves for two species with identical  $L_{\infty}$  of 120cm but differing  $K$  parameters of 0.06 and 0.12. Given the tight relationship between  $L_{\infty}$  and  $L_{\text{mat}}$ , shown in Figures 3 and 4, both species are assumed to have the identical  $L_{\text{mat}}$  of 74 cm. Under these circumstances,  $A_{\text{mat}}$  and  $K$  are seen to be directly inversely proportional; as  $K$  doubles  $L_{\text{mat}}$  halves.

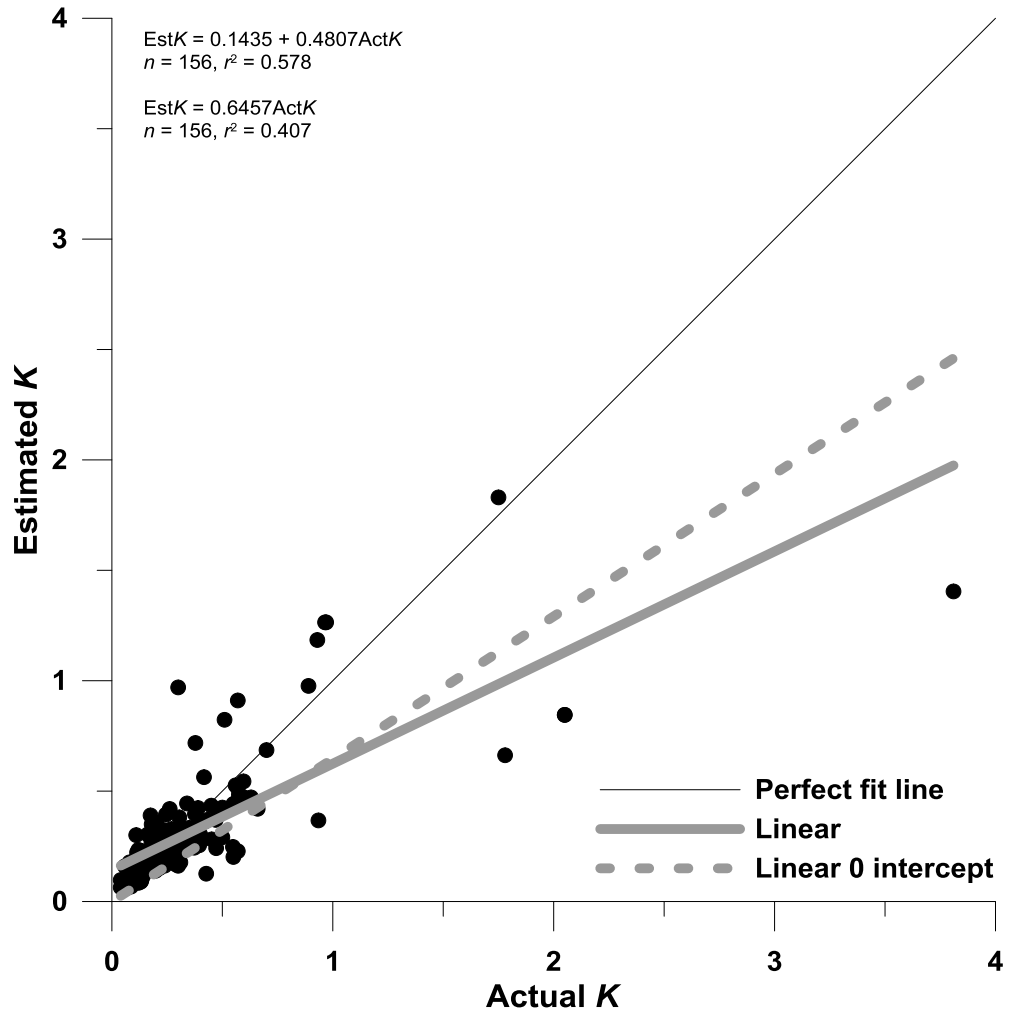


Figure 7.7. Comparison of actual values of K with values of K estimated from a fixed effects GLM of the form  $\ln(K) = \ln(L_{\infty}) + \ln(A_{mat}) + \ln(L_{\infty}) : \ln(A_{mat})$ . The line of perfect prediction is shown as well as two fitted linear functions; one the actual function and the second forced through the origin, as would be expected.



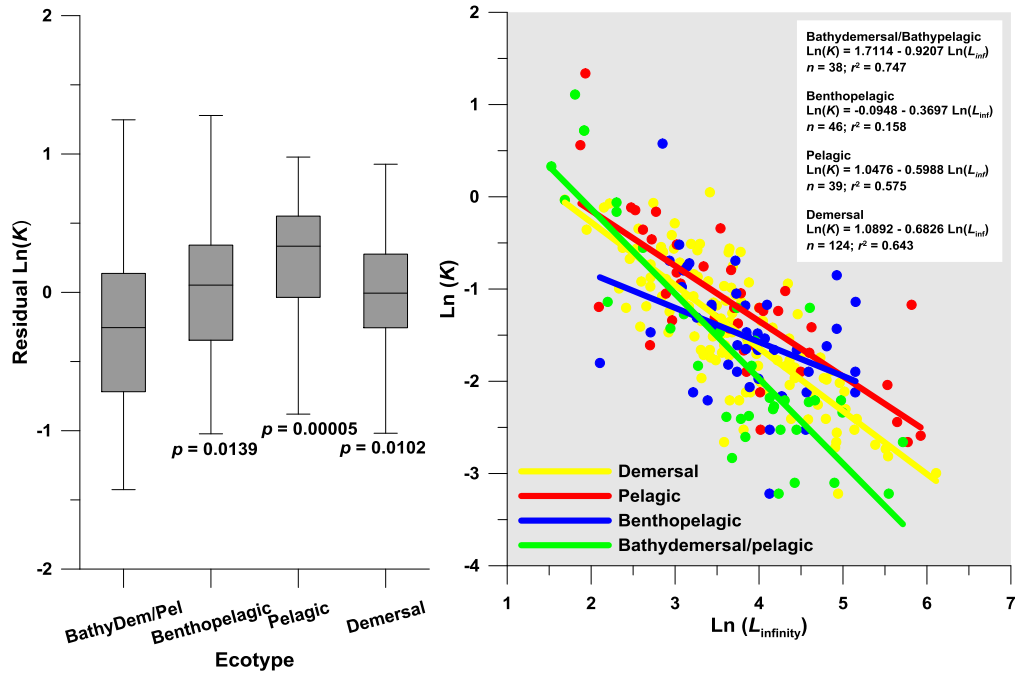


Figure 7.8. Left Panel: Box and Whisker plots showing distributions of the residual variation in the  $L_{\infty}$  -  $K$  relationship for the Bathydemersal and Bathypelagic Ecotypes combined and each of the Benthopelagic, Pelagic and Demersal Ecotypes. Right Panel: Resulting separate models for estimating  $K$  from  $L_{\infty}$  for the combined Bathydemersal and Bathypelagic Ecotypes and each of the Benthopelagic, Pelagic and Demersal Ecotypes.

Where  $A_{mat}$  data were missing, this was estimated directly from the von Bertalanffy growth curve equation. The usual form of the von Bertalanffy equation,  $L_{s,t} = L_{\infty,s}(1 - e^{-K_s(t-t_0)})$ , is used to model how the length of a fish varies over time; knowing the exact age of a fish, its length can be predicted. By setting  $t_0$  to zero and rearranging the equation thus;

$$t = \frac{-\text{Ln}\left(1 - \frac{L_{s,t}}{L_{\infty,s}}\right)}{K_s} \tag{1}$$

the converse can be achieved; knowing a species' length-at-maturity,  $L_{mat,s}$ , and substituting this for  $L_{s,t}$ ,  $t$  then equals  $A_{mat,s}$ . However, by setting  $t_0 = 0$ , these estimates of age include the period of larval development to metamorphosis. This phase of growth is not well characterised by the von Bertalanffy growth equation so its inclusion can distort estimates of age-at-maturity. Gislason *et al.* (2008) assume that fish metamorphose at a length of around 4 cm; an appropriate length for the species they modelled (e.g. Pedersen and Falk-Petersen, 1992), but too large for many of the smaller species sampled in groundfish surveys (Chambers and Leggett, 1987; Benoît *et al.*, 2000; Nikolioudakis *et al.*, 2010). Assuming a link between length at metamorphosis ( $L_{met}$ ) and ultimate body length ( $L_{\infty}$ ), Table 7.2 gives the  $L_{met}$  used here. Substituting these  $L_{met,s}$  values into equation 1 provides estimates of  $t$  that are equivalent to age at metamorphosis,  $A_{met}$ , for each species. Defining age at first maturity as the duration (in years) of the timespan between metamorphosis and first maturation,  $A_{mat,s}$  can be estimated as the difference between these two estimates of  $t$ , thus;

$$A_{mat,s} = \frac{-Ln\left(1 - \frac{L_{mat,s}}{L_{\infty,s}}\right)}{K_s} - \frac{-Ln\left(1 - \frac{L_{met,s}}{L_{\infty,s}}\right)}{K_s} = \frac{-Ln\left(1 - \frac{L_{mat,s}}{L_{\infty,s}}\right) + Ln\left(1 - \frac{L_{met,s}}{L_{\infty,s}}\right)}{K_s} \quad 2.$$

With the exception of one extreme outlier, *Macrourus berglax*, estimates of age-at-maturity derived in this way were a close approximation to actual empirical values for species where age-at-maturity data were available (Figure 7.9).

**Table 7.2. Values of length at metamorphosis,  $L_{met}$ , assumed for species of given ultimate body length,  $L_{\infty}$ .**

LINF (CM)	LMET (CM)
<8	1
8 to <12	1.5
12 to <16	2
16 to <20	2.5
20 to <24	3
24 to <28	3.5
>28	4

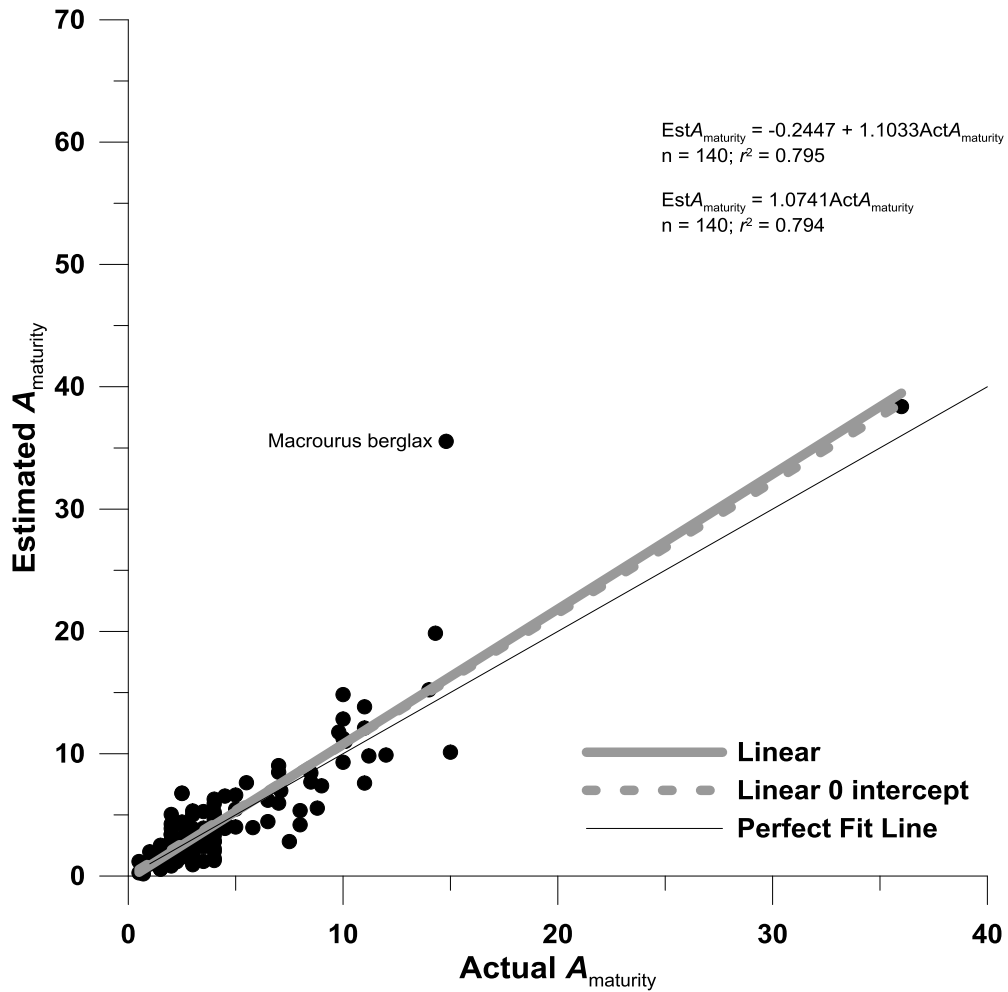


Figure 7.9. Comparison of actual values of  $A_{mat}$  with values of  $A_{mat}$  estimated from the von Bertalanffy growth curve equation. The line of perfect prediction is shown as well as two fitted linear functions; one the actual function and the second forced through the origin, as would be expected.

For just under 50% of species, no von Bertalanffy growth curve parameters are available, so for these species, as illustrated in Figure 1, the estimation of  $L_{\infty}$  from  $L_{MAX}$  is a critical first stage filling missing life-history parameters. Fortunately, the relationship between these two variables is extremely strong, with variation in  $L_{MAX}$  explaining over 91% of variation in  $L_{\infty}$  (Figure 7.10). Again to stabilise the variation in residuals, the ln-transformed data were used. Despite this tight relationship, GLMs revealed a significant effect of ‘Class’ (Figure 7.11: left panel), so that once again, two separate models could be used to estimate  $L_{\infty}$  from  $L_{MAX}$  (Figure 7.11: right panel).

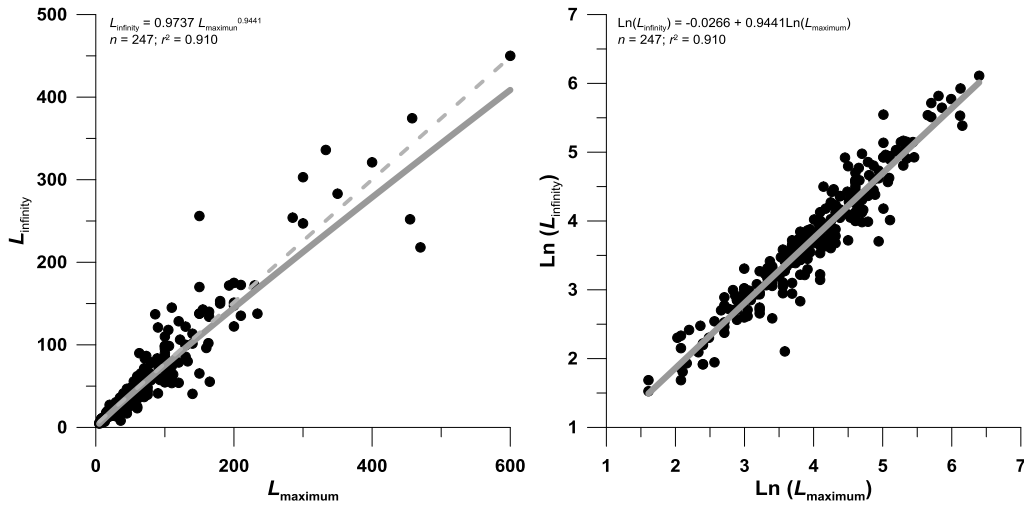


Figure 7.10. The relationship between  $L_{\infty}$  and  $L_{MAX}$ . Left panel shows the untransformed data and the power function equivalent to the linear relationship fitted to the natural log-transformed data in the right panel.

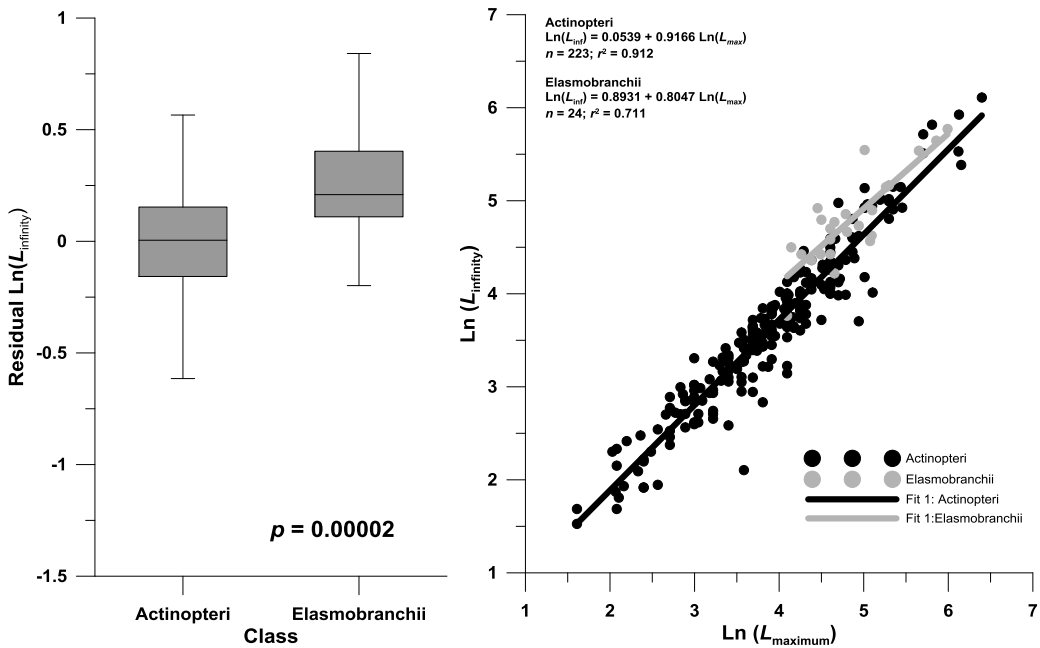


Figure 7.11. Left Panel: Box and Whisker plots showing distributions of the residual variation in the  $L_{\infty} - L_{MAX}$  relationship for Actinopteri and the other classes combined and for Elasmobranchii. Right Panel: Resulting separate models for estimating  $L_{\infty}$  from  $L_{MAX}$  for Actinopteri and the other classes combined and for Elasmobranchii.

Once three of the four parameter values were known, either from empirical data or modelled estimates, a value for the fourth parameter could always be derived from the von Bertalanffy growth curve equation. Thus, just as  $A_{mat}$  can be derived from the von Bertalanffy Growth Curve where  $L_{\infty}$ ,  $K$  and  $L_{mat}$  are known;

$$A_{mat} = \frac{-\ln\left(1 - \left(\frac{L_{mat}}{L_{\infty}}\right)\right)}{K} - A_{met}, \tag{3}$$

so  $L_{mat}$  can be determined if  $L_{\infty}$ ,  $K$  and  $A_{mat}$

$$L_{mat} = L_{inf} \left( 1 - e^{-K(A_{mat} - A_{mst})} \right) \tag{4}$$

and  $K$  can be determined if  $L_{\infty}$ ,  $L_{mat}$  and  $A_{mat}$  are known

$$K = \frac{-\ln \left( 1 - \left( \frac{L_{mat}}{L_{inf}} \right) \right)}{A_{mat} - A_{mst}} \tag{5}$$

The parameter estimation protocols used to address each of the seven missing parameter value scenarios portrayed in Table 7.1 are given in Figure 7.12, and Table 7.3 lists each of the model classes, the parameter each model was used to estimate, and the number of different types of situation in which it was used. This latter information links to the protocols listed in Figure 7.12. Once three of the four parameter values were known, either from empirical data or modelled estimates, a value for the fourth parameter could always be derived from the von Bertalanffy growth curve equation. Table 7.4 gives the actual numerical models used to estimate missing values.

### Total of 469 species

$L_{inf}$	$K$	$L_{mat}$	$A_{mat}$	$N$	
😊	😊	😊	😊	140	
😊	😊	😊	❌	45	$A_{mat} = \text{VBGC: } L_{inf}, K, \text{ and } L_{mat}$
😊	😊	❌	😊	16	$L_{mat} = \text{VBGC: } L_{inf}, K, \text{ and } A_{mat}$
😊	😊	❌	❌	46	Step 1 - $L_{mat} = \text{GLM(2): } L_{inf} \text{ and Class}$ Step 2 - $A_{mat} = \text{VBGC: } L_{inf}, K, \text{ and } L_{mat}$
❌	❌	😊	😊	7	Step 1 - $L_{inf} = \text{GLM(2): } L_{max} \text{ and Class}$ Step 2 - $K = \text{VBGC: } L_{inf}, A_{mat}, \text{ and } L_{mat}$
❌	❌	😊	❌	36	Step 1 - $L_{inf} = \text{GLM(2): } L_{max} \text{ and Class}$ Step 2 - $K = \text{GLM(4): } L_{inf} \text{ and Ecotype}$ Step 3 - $A_{mat} = \text{VBGC: } L_{inf}, K, \text{ and } L_{mat}$
❌	❌	❌	😊	5	Step 1 - $L_{inf} = \text{GLM(2): } L_{max} \text{ and Class}$ Step 2 - $K = \text{GLM(4): } L_{inf} \text{ and Ecotype}$ Step 3 - $L_{mat} = \text{GLM(2): } L_{inf} \text{ and Class}$
❌	❌	❌	❌	174	Step 4 - $A_{mat} = \text{VBGC: } L_{inf}, K, \text{ and } L_{mat}$

Figure 7.12. Parameter estimation protocols applied to address each missing life-history parameter situation.

Table 7.3. List of the different model classes used to estimate missing life-history trait parameters, the parameter they estimated, and the number of different situations when each model was used (see Figure 12).  $A_{mat}$  can be derived from the von Bertalanffy Growth Curve where  $L_{\infty}$ ,  $K$  and  $L_{mat}$  are.

Missing Parameter	Model	Number of Times Used
$L_{infinite}$	GLM(2): $L_{max}$ and Class	4
$K$	GLM(1): $L_{inf}$ and $A_{mat}$	1
$K$	GLM(4): $L_{inf}$ and Ecotype	2
$K$	VBGC: $L_{inf}$ , $A_{mat}$ , and $L_{mat}$	1
$L_{maturity}$	GLM(2): $L_{inf}$ and Class	2
$L_{maturity}$	VBGC: $L_{inf}$ , $K$ , and $A_{mat}$	2
$A_{maturity}$	VBGC: $L_{inf}$ , $K$ , and $L_{mat}$	4

Table 7.4. Mathematical equations used to derive missing parameter values.

PARAMETER	CATEGORICAL FACTOR	MATHEMATICAL EQUATION
$L_{\infty}$	Actinopteri +	$\ln(L_{\infty})=0.0539+0.9166\ln(L_{max})$
$L_{\infty}$	Elasmobranchii	$\ln(L_{\infty})=0.8931+0.8047\ln(L_{max})$
$K$	All	$\ln(K)=1.1708-0.5552 \ln(L_{\infty})-0.8755\ln(A_{mat})+0.0133(\ln(L_{\infty})^* \ln(A_{mat}))$
$K$	Bathydem/pel	$\ln(K)=1.7114-0.9207\ln(L_{\infty})$
$K$	Bentho-pelagic	$\ln(K)=-0.0948-0.3697\ln(L_{\infty})$
$K$	Pelagic	$\ln(K)=1.0476-0.5988\ln(L_{\infty})$
$K$	Demersal	$\ln(K)=1.0892-0.6826\ln(L_{\infty})$
$K$	All	$K=-\ln(1-(L_{mat}/L_{\infty})) / (A_{mat}-A_{met})$
$L_{mat}$	Actinopteri +	$\ln(L_{mat})=-0.2523+0.8983\ln(L_{\infty})$
$L_{mat}$	Elasmobranchii	$\ln(L_{mat})=-0.2819+0.9570\ln(L_{\infty})$
$L_{mat}$	All	$L_{mat}=L_{\infty}(1-e^{-K(A_{mat}-A_{met})})$
$A_{mat}$	All	$A_{mat}=(-\ln(1-(L_{mat}/L_{\infty})) / K) - A_{met}$

## 7.2 Deriving the Greenstreet et al. (2012) 'sensitivity' metric

For  $L_{\infty}$ ,  $L_{mat}$ , and  $A_{mat}$ , increasing values corresponded to increased sensitivity, but since  $K$  is negatively correlated with  $L_{\infty}$  (Figure 1), the reverse was true; increased sensitivity was associated with decreasing values of  $K$ . By taking reciprocals of  $K$  the direction of this relationship was reversed, ensuring that variation in all four life-history trait variables was positively related to sensitivity. Having determined values for each of the four life-history trait variables for all 469 species, these values were square root transformed to reduce the spread in each set of data and to distribute the values more evenly within their ranges. The data for each variable were then standardised following

$$\Omega_{std,s} = \frac{\Omega_s - \Omega_{min}}{\Omega_{max} - \Omega_{min}}$$

where  $\Omega_s$  is the value for any of the four life-history trait variables for a particular species,  $\Omega_{\min}$  is the minimum and  $\Omega_{\max}$  is the maximum value for that life-history variable observed among the 469 species.  $\Omega_{\text{std},s}$  is the standardised value for that life-history trait variable for the species in question. A single sensitivity index for each of the 469 species was then derived simply by determining the arithmetic mean of the four standardised life-history trait variables. This index ranged between 0 representing the least sensitive, or most resilient, species and one representing the most sensitive species.

Figure 7.13 shows how the ‘sensitivity’ metric varies from species to species when the species are ranked in order from least ‘sensitive’ to most ‘sensitive’. The figure shows the upper and lower 33%iles used previously to define ‘sensitive’ and ‘resilient’ species respectively. An alternative method for defining ‘sensitive’ species would be to fit a 5th degree polynomial. A high degree function was used to obtain the closest possible fit to the curve and especially to capture any inflexion point present. The point where the gradient started to become increasingly positive towards the high ‘sensitivity’ metric range of the plot was chosen as the point where species should become defined as sensitive. The figure legend provides further details. Species ranked 332 was thus defined as the first ‘sensitive’ species, giving a total of 138 sensitive species, 29.4% of the total of 469 species.

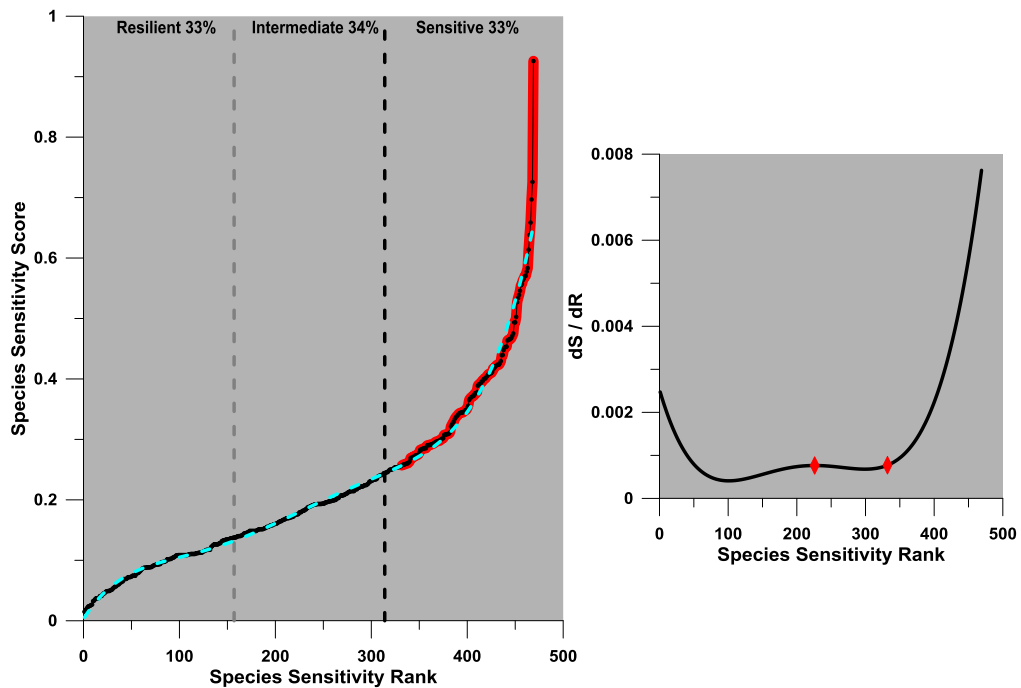


Figure 7.13. Left panel shows variation in the ‘sensitivity’ metric determined following the approach advocated by Greenstreet *et al.* (2012). Plot shows variation in metric value across species ranked from least ‘sensitive’ to most ‘sensitive’ (black line). The upper (defining ‘sensitive’ species) and lower (defining ‘resilient’ species) 33%iles are indicated. Blue dashed line shows the 5th degree polynomial function fitted to the data. Right panel shows the 1st order differential of the fitted polynomial function and marks the two points used to define the point beyond which species are considered ‘sensitive’. Left diamond marks the position of the highest positive gradient in the ‘non-sensitive’ species range. Right diamond marks the species rank of 332, where the positive gradient of the curve first exceeds the gradient at the position of the left diamond. Species identified as ‘sensitive’ following this procedure are highlighted by the red fraction of the curve.

### 7.3 A 'sensitivity' metric that takes parameter interactions into account

Considering fishing to be historically and probably still currently the major source of additional mortality, then several points emerge. First, fishing is size selective, driven primarily by the minimum landing sizes and mesh sizes imposed through fisheries management legislation. Clear size selectivity is evident from the catchability coefficients for fisheries research survey trawls gears (Fraser *et al.*, 2007). This implies that only at a certain size do fish become vulnerable to fishing pressure and start to suffer a fishing mortality term. Based on the fishing regulations currently enforced by the EC, modelled and observed data (Piet *et al.*, 2009), a length at which fish start to become exposed to additional mortality,  $L_{exp}$ , of 25 cm is assumed. The time, or age at which fish of each species start to experience fishing mortality can therefore be estimated as follows,

$$A_{exp} = \left( \frac{-\text{Ln} \left( 1 - \frac{L_{exp}}{L_{inf}} \right)}{K} \right) - A_{mat} \quad 7.$$

A fraction of fish surviving to reproduce is a key element in determining each species' sensitivity (Gislason *et al.*, 2008; Le Quesne and Jennings, 2012). Clearly therefore, the length of the period between when a fish starts to be exposed to fishing pressure and when it first spawns is a critical factor in determining the fraction of each cohort surviving to reproduce. Equation 7 shows the rearrangement of the von Bertalanffy growth curve equation used to estimate  $A_{exp}$ . The length of the period of exposure to fishing mortality,  $T_{fish}$ , can therefore be determined as,

$$T_{exp} = A_{mat} - A_{exp} \quad 8.$$

From these equations it is clear that only three of the four principal life-history parameters are needed. This takes account of the fact that the von Bertalanffy Growth Curve equation was always used to determine the last missing parameter value. Since the equations above are all rearrangements of the von Bertalanffy equation, use of the fourth parameter would provide no additional information. The  $A_{mat}$  term was, however, still needed to correct for the poor fit of the von Bertalanffy equation to the larval growth phase. Clearly fish with a  $L_{inf}$  lower than  $L_{fish}$  should rarely grow to a size where it is experiencing mortality from fishing, and fish with  $L_{mat}$  lower than  $L_{fish}$  should all survive to spawn at least once. In the former instance, equation 7 becomes insolvable because the term  $L_{fish}/L_{inf}$  becomes  $>1$ . In both cases the 'sensitivity' metric defaults to a value of 1; additional mortality associated with anthropogenic activities causes no reduction in the fraction of a cohort surviving to spawn. To determine the 'sensitivity' metric, 10 000 fish of each species were assumed to reach  $L_{fish}$  at time  $A_{fish}$  and the number ( $N_{mat}$ ) of these surviving to spawn at  $L_{mat}$  and  $A_{mat}$  was determined using a simple mortality rate equation,

$$N_{mat} = 10000e^{-zT_{fish}} \quad 9.$$

Figure 14 shows how this new 'sensitivity' metric varies from species to species when the species are ranked in order from least 'sensitive' to most 'sensitive'. This again poses the question as to how set the threshold to define sensitive species. Here we have adopted an L50 approach; a species is considered 'sensitive' if exposure to additional anthropogenic sourced mortality reduces the number of fish surviving to spawn by 50%. But now additional issues arise, the 50% survival fraction is depend-



ent both on the mortality rate assumed and the length/age at which exposure to the additional mortality commences. The effect of the former is illustrated in Figure 7.14. As mortality rate increases, curves move to the left; species become classified as 'sensitive' at a lower rank, increasing the number of 'sensitive' species. The effect of the latter has still to be examined. However, if exposure to mortality commences earlier and a lower length, then the curves shown in Figure 7.14 should again simply move to the left; for any given mortality rate, species would be classified as 'sensitive' at a lower rank, again increasing the number of 'sensitive' species. Figure 7.14 also shows the trend in the period of exposure and this could also be adopted as the 'sensitivity' index. However, this metric suffers the same problems as the Greenstreet *et al.* (2012) metric as again there is no obvious threshold for defining 'sensitive' fish. It is however, unaffected by choice of mortality rate, and but choice of length at which species become exposed to additional mortality will affect the metric.

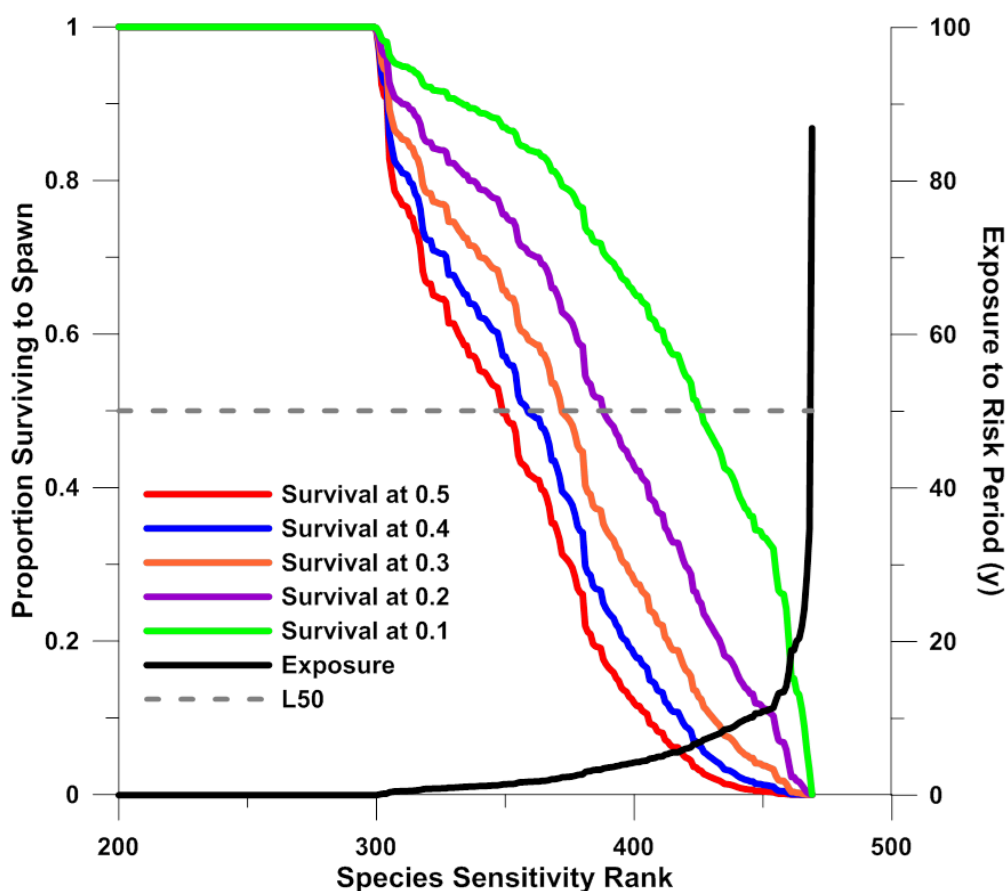


Figure 7.14. Variation in the von Bertalanffy derived 'sensitivity' metric across 469 species sampled in Northeast Atlantic groundfish surveys ranked from least 'sensitive' to most 'sensitive'. Below a rank of 200, all species have the default 'sensitivity' score of 1. The effect of mortality rate choice is demonstrated. Variation in the period of exposure ( $T_{exp}$ ) is also shown.

Here an additional mortality rate of 0.5 is chosen. At this mortality rate, the species at rank 349 was the first to have a survivorship of <50%, marking this is the lowest ranked 'sensitive' species and giving a total of 121 'sensitive' species.

#### 7.4 Evaluating the data paucity issue

Both sensitivity metrics described here rely on the availability of four separate life-history trait parameters for each species considered. Here metrics have been derived for 469 species in groundfish surveys carried out across the Northeast Atlantic. Table 7.1 illustrates the extent of data availability. Turning this point around, it can be determined from Table 7.1 that the 469 species require a total of 1876 life-history parameter values, but in 986 instances (52.6%) the required data are absent. To derive the 'sensitivity' therefore puts considerable emphasis on the need for reliable models with which to estimate all this missing data. The analyses presented suggest that the ultimate size to which a species grows has a fundamental influence on its sensitivity to any additional mortality associated with anthropogenic activities. Where  $L_{inf}$  data are missing this is always modelled using  $L_{MAX}$  data, because  $L_{MAX}$  data are available for every species in the inventory; begging the question as to whether each species  $L_{MAX}$  value alone might not provide an adequate indicator of its sensitivity to additional mortality?

To address this question species were ranked according to their  $L_{MAX}$  and these rankings compared with those for the two sensitivity metrics. The key issue was to determine the extent to which using a species  $L_{MAX}$  resulted in species classified being 'sensitive' by the 'sensitivity' metric were classified as resilient on their  $L_{MAX}$  ranking. Figure 7.15 demonstrates that using  $L_{MAX}$  alone to determine each species 'sensitivity' results in an appreciable number of species classified as 'sensitive' being classified as resilient on the basis of  $L_{MAX}$  alone. The figure also suggests that this mismatch is greater when 'sensitivity' is determined using the metric based on the von Bertalanffy growth curve.

		Average Metric	
		Sensitive	Resilient
Lmax	Sensitive	120	18
	Resilient	18	313
		13%	
		VBGC Metric	
		Sensitive	Resilient
Lmax	Sensitive	97	24
	Resilient	24	324
		17%	

Figure 7.15. Mismatching in the identification of species identified as ‘sensitive’ by either of the ‘sensitivity’ metrics, being classified as “resilient” if L<sub>MAX</sub> alone were used.

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**8 Consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level (in collaboration with the DEVOTES project and building on work from WGBIOV) (ToR f) and the request ‘In the light of previous advice, review the guidance on the most suitable and defensible approach to aggregate species within species groups (such as birds, mammals, reptiles, fish and cephalopods), for the state assessments of the MSFD that was developed by the Workshop on providing a method to aggregate species within species groups for the assessment of GES for MSFD D1 (WKD1Agg)’**

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It is recognized that the advice request was for D1 only, however, WGECO considered that the integration approaches should also be appropriate to other descriptors, especially D3, and probably also D4 and D6. WGECO have elaborated on the links to D3 in the following section on ToR F. Therefore, WGECO both reviewed the WKD1Agg report and then proceeded to consider the recommendations on integration given by WKD1Agg and WKGESFish together with WGECO considerations.

**8.1 Review the guidance developed by the Workshop on providing a method to aggregate species within species groups for the assessment of GES for MSFD D1 (WKD1Agg)**

WGECO reviewed the draft report of WKD1Agg principally in terms of the value of the report in providing advice for MSFD managers and of its consistency with previous recommendations from WGECO on integrating MSFD indicators. WGECO did not have access to the revised MSFD.

The report was considered as very useful, and covered the main elements needed to combine indicators.

Before drafting the advice in response to the request a number of important issues need to be considered:

- The section about integration methods is incomplete, for example it lacks the fuzzy operators which formalize a wide flexibility in integration methods (Silvert, 1997; 2000). The section about averages also fails to mention geometric means, whereas it has been shown to be generally more relevant than arithmetic mean for indicator integration (Ebert and Welsh, 2004). The section about integration methods should be structured according to the methods which integrate quantitative indicators vs. those which integrate assessments. Quantitative indicators may be aggregated by e.g. averages and weighted averages, which are essentially the same (averages being weighted averages with equal weights). By contrast, the percentage of indicators within limits (of which the “one out all out” rule is just a special case, with the required proportion 100%) integrates assessments resulting from combinations of indicators with the associated reference point, i.e. a qualitative statement such as “Good environmental status” or “Outside safe biological limits”. The qualitative assessments require the availability of thresholds established in a consistent way for all the indicators to be integrated.

- A corollary is that averaging cannot be used after an assessment has been made at a lower level; which disqualifies the example in Section 5.1.4.4, Figure 10, where proportions are used at the lower level, preventing any averaging to be used at the next level.
- In a number of examples e.g. 4.4 and 4.5 (averages and weighted averages), the text mentions the idea of indicators “compensating” for each other. This terminology is confusing. At a simple level it could mean that a bad GES in one indicator, could be “compensated” by a good one in another. However, the text also suggests that indicators may or may not have the “ability to compensate for each other”. This implies a judgement, and would be very difficult to substantiate. We would suggest the use of the term “cancel out” and make it explicit that this means that having one good, and one bad indicator in an integration step could lead to them “cancelling” each other out.
- It was quite difficult to identify the “cons” in the “pros and cons” sections. The general practice appeared to be to give the advantages, and then suggest a variety of approaches that might mitigate the “cons” that were not detailed clearly. We would suggest that the advice could include bulleted lists for the pros and cons, and then possibly suggest the mitigation of the cons specific to each of these. A table of method and pros and cons like the one given in Section 8.2 of this report might be useful.
- In many cases, the text discussed the problems of producing a single result from the integration at any level might “cover up” GES issues, at the lower level. This would apply to any of the averaging approaches. The text also differentiates between indicators that are below but close to a “good” status, from those very far from being “good”. In all such cases we would recommend that, whatever the outcome of the integration process, the reporting should also detail where such cases had arisen. This would be vital information to target remediation or research.
- At the higher levels of integration the text implies that after an OOA process, the components or components causing that “out” should be identified. As with the averaging discussed in the previous bullet, these should include how bad the components causing the “out” evaluation actually were.
- On page 7 and thereafter the text uses the term “areas” as distinct from “regions” and subregions” detailed in Table 3. We understand these “areas” to be specific to the management areas in the Baltic Sea where a slightly different approach was proposed. However, this should be made clear, and it should be questioned whether a Baltic specific step in the integration is useful.
- The report explicitly avoids considering “pressure indicators”, based on the specific request for advice. WGEKO believes that any useful integration process should be able to encompass both pressure and state indicators at a disaggregated level.
- Many indicators so far proposed can be considered as “surveillance”, rather than “operational”, indicators. It should be made clear that this advice refers to operational indicators only, and probably only those where reference levels or some other good/bad state threshold can be identified.

- WGECO noted that there was some attempt to harmonise the D1 indicator approach with that used for the Habitats Directive. While this is not a problem in itself, it was felt that harmonisation should probably be emphasised first within the MSFD and only then across directives.
- The diagrams were felt to be valuable and helped in explaining both the principles and how it might work in action. However, if these are to be used in the advice, there are some quality and consistency issues that need to be addressed. For instance there is virtually no difference between Figures 2 and 3, whereas they are supposed to illustrate two contrasted aggregation processes. In Figures 8, 9, 11 and 12 there was no indication of an integration process at the lowest level.

### Minor points

- Section 2. The information referred to from Palialexis *et al.* should probably be included.
- Section 3 paragraph 1. End of the sentence should probably include “in abundance.”
- Figure 6 is very poor quality, and impossible to see the elements indicated in the legend.
- Section 4.5. Paragraph 2. Line 2. This seems to be conflating more data with better data and mixing up quality and quantity of data. It is probably about quality of data.
- Section 5.1. Final paragraph. It is not clear what an “unacceptable deviation from the good status means” unless it simply refers to good vs. Bad??

There have been several ICES workshops that considered the issue of aggregation/integration in relation to an assessment of status of an MSFD Descriptor:

- For D1 the WKD1Agg the ToR was “In the light of previous advice, provide guidance on the most suitable and defensible approach to aggregate species within species groups (such as birds, mammals, reptiles, fish and cephalopods), for the state assessments of the MSFD.”
- For D3 the WKGESFish the ToR was: “Conduct the assessment of criterion 3.1, 3.2 and 3.3, to evaluate the GES status of selected stocks (as examples),” which specifically involved an exploration of the methods for integrating indicator assessment results within stocks across Criteria 3.1 and 3.2, as well as the aggregation of stocks within criteria.

WGECO considers that the methods applied to assess the status of MSFD descriptors should not be considered in isolation for each of the descriptors separately but also across descriptors as this is likely to increase consistency in the assessments and thus comparability of the outcome of the assessments which is an advantage if trade-offs between the descriptors need to be considered.



## 8.2 Consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level; comparing WKD1Agg, WKGESFish and WGEKO approaches

The WKD1Agg concluded that there were two feasible integration frameworks relevant to the assessment of D1: the integration within criteria and integration within species. This was also found by WKGESFish which, as an aid to compare and discuss these different frameworks, proposed to distinguish between the words ‘aggregation’ and ‘integration’ which are often used synonymously but can carry different connotations, i.e.

- ‘aggregation’ refers to the combination of several elements which are similar (e.g. the aggregation of the same indicator across species or stocks or aggregation of species within the same criterion), whereas
- ‘integration’ refers to the synthesis of several elements, which are not comparable (e.g. the integration of several indicators or criteria within a stock, species or species group) (Borja *et al.*, 2014).

Below we discuss the conclusions from WKD1Agg together with those coming from WKGESFish and considerations by WGEKO. It should be noted that the two methods are identical if One Out All Out methods are used for integration.

WGEKO also questioned the relevance for management of highly aggregated or integrated evaluations that are effectively averages of a wide and varied set of indicators. Averaging up to an ecosystem component or to a species group, and then evaluating that component/group as being in or out of GES would tend to disguise the main issues. For components/groups that are in GES, managers and stakeholders would presumably need to identify those elements that were still outside GES and to target remedial action. The same would be true if the overall evaluation were to be outside GES; the managers would still need to know which specific indicators were driving this. It would also be useful to know by how much any given indicator was outside GES. Arguably, managers may want to focus on those closest to reaching their targets, although this may be “picking low hanging fruit”. A better approach might be to carry out an ecological risk assessment (e.g. Hobday *et al.*, 2011), and identify the most important elements outside GES. The key point is that these high level integration results may have some political value, but be of little use for targeting attention or action where it would be needed.

INTEGRATION WITHIN SPECIES		
D1 (WKD1Agg)	D3 (WKGESFish)	D1, D3, D4 and D6 (WGEKO)
(+) Increases comparability with Habitats Directive	(-) Decreases comparability with Common Fisheries Policy	(-) decreases comparability with D1-habitat, D4 and D6, neither of which are species based
(-) Community considerations are not easily integrated		(-) Assessment of D1-habitat, D4 and D6 is not likely to depend on species-specific indicators but rather indicators involving several species into a single indicator (e.g. biomass or production of a trophic guild)
(-) When different species have information for different numbers of criteria, this is masked in the higher-level integration.	(-) There are many stocks with information on only one of the criteria. If the requirement is that only stocks with both criteria can be included a lot of information is lost. Alternatively, allowing also stocks with only one criterion could be an incentive to provide only one (best) criterion for each stock. In both cases the outcome of the assessment will be affected.	
	(-) Proposes to retain the focus on stocks rather than species, as this promotes coherence with the CFP (which considers stocks) this would therefore require having to aggregate stocks into species.	

AGGREGATION WITHIN CRITERIA		
D1: WKD1Agg	D3: WKGESFish	WGECO
(-) Decreases comparability with Habitats Directive	(+) Increases comparability with Common Fisheries Policy	
(+) Data for species where only one criterion can be measured can be included without loss of consistency and all criteria receive equal weight in the integration across criteria.	(+) Avoids the problems associated with combining indicators with different evidence base and levels of confidence (i.e. primary vs. secondary indicators), as well as the conceptual problem of combining pressure-related indicators (3.1) with state-related indicators (3.2).	(+) Retains the equal importance of all criteria, emphasizing that no one criterion is considered more important than any other criterion.
	(+) Proposes to retain the focus on stocks rather than species, as this promotes coherence with the CFP. This therefore avoids having to aggregate stocks into species.	(+) Combining stocks into species or criteria can be done either assuming stocks to be equally important or by assuming stocks to differ in importance. The current implementation of the CFP uses the former method but using an agreed weighting, such as biomass or areal coverage, may be more appropriate to e.g. foodweb considerations.
(+) Transparent weighting of all criteria, including community aspects when carrying out the next level of integration/aggregation.		(+) Transparent weighting of all criteria when carrying out the next level of integration/aggregation.
(+) More appropriate where there is a larger number of species, due to the increasing chance of some species missing information on one or more criteria with an increasing number of species	(+) Because D3 assessment usually involves large number of fish species, this is considered most appropriate method	(+) More appropriate where there is a larger number of indicators in accordance with the advice for D4 to monitor at least three trophic guilds.

INTEGRATION/AGGREGATION METHODS		
D1 (WKD1Agg)	D3 (WKGESFish)	D1, D3, D4 and D6 (WGECO)
<p>The recommended integration method differs between integration levels under both of the two frameworks. At the level of integrating criteria, the recommended method is OOAO, and the same recommendation applies when aggregating species groups. For aggregation across species a mixture of averages, weighted averages, proportional and probabilistic methods are recommended depending on the specific situation.</p>	<p>At the level of integrating criteria, the recommended method is OOAO. The focus was mainly on two aggregations methods, the One-Out-All-Out approach (OOAO) and averaging methods. Probabilistic methods were only briefly discussed during the meeting, but were deemed to hold high potential for the determination of GES threshold levels for the aggregation of stocks within criteria.</p>	<p>At the level of integrating criteria, the recommended method is OOAO. For aggregation across indicators within a criteria a mixture of averages, weighted averages, proportional and probabilistic methods are recommended depending on the specific situation.</p>

CONCLUSIONS		
D1: WKD1Agg	D3: WKGESFish	WGECO
<p>The final choice of integration framework depends on the relative consideration of the wish to facilitate comparability with the Habitats Directive (integration within species), the wish to use similar methods across all ecosystem components, the wish to give a transparent weight to all criteria (integration within criteria) vs. giving transparent weight to all species (integration within species) and finally, the wish to incorporate community-level considerations in the integration in a simple way. The workshop participants considered that the weighing of these different wishes was a policy decision rather than a scientific decision.</p>	<p>The aggregation of stocks within criteria was considered as preferable by most workshop participants, because the aggregation of stocks within criteria would avoid the problems associated with combining indicators with differing evidence base and levels of confidence (primary vs. secondary indicators), as well as the conceptual problem of combining pressure-related indicators (3.1) with state-related indicators (3.2). The aggregation of stocks within criteria would also have the advantage that the available information would be optimally used and that results match those for the CFP.</p>	<p>Comparability across methods makes the final assessment at descriptor level more transparent and facilitates communication. There does not appear to be a good argument for using species approach in one descriptor but criteria approach in another.</p>

### 8.3 References

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## Annex 1: List of participants

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## Annex 2: Agenda

TORS REFERRED TO ARE GIVEN IN THE BOTTOM OF THE DOCUMENT		
Wednesday April 6th		
1000	Opening of the meeting	
	Adoption of ToR & Agenda	
	Overview of volunteers to work on different ToRs and overview of presentations prepared for the meeting	
	Initial discussion of ToRs f and g) Consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level (tor f) or ecosystem component level of D1 (tor g, review of WKD1Agg);	
1300	Lunch	
1400	Reconvene. Initial discussions on ToR c) Use the data available to evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are "balanced". Please see the details from Jeremy at the bottom of the document on the kind of data needed from different systems.	
1600	Coffee	
1630	Initial discussions on ToR d)	Distributional indicators in the context of MSFD: developing operational and surveillance indicators and exploring the link to pressure and drivers (this tor may be supplemented by a request to point to the most appropriate indicators to detect distributional change) and e)
	Review and further develop indicators of state of sensitive species throughout the ICES area and suggest risk based advice on sensitive species	
1730	Develop Following Day Workplan, Updates from earlier in day, Wrap up	
1800	Adjourn	



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Thursday April 7th

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900	Initial discussion of ToR a) Develop and assess indicators of Good Environmental Status for seabed integrity: Further develop indices of impact on the seabed and sensitivity of the benthic community to different pressures, and the link to function (joint ToR with BEWG) and b) Investigate possible indicators of scavengers, examine their relation to discard amounts and evaluate the spatial effect of a landing obligation on the scavengers
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1030	Coffee
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1300	Lunch
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1400	Subgroups on ToRs c, a/b, f/g and d/e
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1600	Coffee
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1800	Adjourn
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Friday April 8th

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900	Plenary status update from all subgroups
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1030	Coffee
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	Subgroups on ToRs c, a/b, f/g and d/e
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1300	Lunch
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1400	Reconvene
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	Subgroups on ToRs c, a/b, f/g and d/e
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1600	Coffee
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	Plenary status update from all subgroups
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1730	Develop Following Day Workplan, Updates from earlier in day, Wrap up
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1800	Adjourn
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Saturday April 9th

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900	Plenary: Preliminary check on WG report elements
	Revisit, as need be, ToRs c, a/b, f/g and d/e
1030	Coffee
	Subgroups on ToRs c, a/b, f/g and d/e
1300	Lunch
1400	Subgroups on ToRs c, a/b, f/g and d/e
1600	Coffee
1630	Plenary status update from all subgroups
	Develop Following Day Workplan, Updates from earlier in day, Wrap up
1800	Adjourn

Sunday April 10th

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Writing day, no plenaries

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Monday April 11th

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900	Subgroup reporting ToRs a–g
1030	Coffee
	Drafting session
1300	Lunch
1400	Reconvene
	Drafting session
1600	Coffee
1800	Adjourn

Tuesday April 12th

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900	Subgroup reporting
1030	Coffee
	Drafting session
1300	Lunch
1400	Reconvene
	Drafting session
1600	Coffee
1800	Adjourn

Wednesday April 13th

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900	Scope out next year's meeting plan, schedule, ToR, election of chair for WGEKO 2017–2019
1030	Coffee
	Tying up loose ends
1300	Adjourn

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### **Annex 3: WGECO terms of reference for the next meeting**

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The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by Jeremy Collie (US) and Stefan Ragnarsson (Iceland), will meet in Reykjavik, Iceland 5–12 April 2017 to:

- a) Integrate large-scale maps of sensitive benthos and fish and relate this to spatial distribution of effort and landings to identify high-fishing-low-sensitivity areas and low-fishing-high-sensitivity areas:
  - i) Request VMS effort maps where these are not available;
  - ii) Rectangle based catches for the species listed;
  - iii) Request map of sensitive habitats where these are available from WKFBI or BEWG.
- b) Complete the investigation of possible indicators of scavengers, examine their relation to discard amounts and evaluate the spatial effect of a landing obligation on the scavengers;
- c) Use the data available to evaluate the degree to which fisheries in the ICES region are “balanced”:
  - i) establish the distribution of total catch (landing+discards) among size classes (catch size spectrum), species and functional groups;
  - ii) Examine how the degree of balance is related to ecosystem status;
  - iii) Request catch by species and length group for the species listed;
  - iv) Request survey biomass by species and length group, where possible catchability corrected.
- d) Estimate indicators of state of sensitive fish species throughout the ICES area;
- e) In support of providing ecosystem advice, define a list of relevant pressure, driver and state indicators to be estimated by relevant experts groups, including stock assessment groups.

WGECO will report by 24 April 2017 to the attention of the Advisory Committee.

ToRs for consideration for the 2017 meeting:

- i) Distributional indicators in the context of MSFD: developing operational and surveillance indicators and exploring the link to pressure and drivers;
- ii) Ecosystem effects of trawling (1st and 2nd order effects will draw together work on sensitive fish and benthos, distribution, dependent predators, affected prey);
- iii) Determine where ecosystem effects of rebuilding predators are likely to occur;
- iv) Expanding the DPSIR framework with links to ecosystem services: How does state and impact relate to the provision of ecosystem services and human wellbeing;
- v) Continue the development of foodweb indicators and comment on the suggested foodweb indicators from WKFooWI and WGSAM;

- vi) Identify data weaknesses in the data available to address WGEKO ToRs and recommend priority areas for data collection and model development.

### Supporting Information

Priority	<p>The current activities of this Group will lead ICES into issues related to the ecosystem affects of fisheries, especially with regard to the application of the Precautionary Approach. Consequently, these activities are considered to have a very high priority.</p>
Scientific justification	<p>Term of Reference a)  The integration of different ecosystem considerations before suggesting management measures is key to achieving beneficial effects on all ecosystem aspects rather than opposing effects on different ecosystem components. Further, once potential sensitive areas have been identified, consideration of socio-economic aspects require that it is at least considered how ecological aspects can be maintained at the lowest possible socio-economic cost.</p> <p>Term of Reference b)  The implementation of a landing obligation is likely to result in major changes to the impact of fishing on particularly benthic scavenging species. The magnitude of both direct and indirect effects are unknown at present and further work is needed in order to provide advice on the direction and magnitude of change as a result of the landing obligation.</p> <p>Term of Reference c)  The overall objective of this ToR is to determine whether a balanced fishery imposes a smaller impact on the ecosystem. Determining the ecological consequences of the degree to which fisheries are “balanced” ideally requires a large metadataset to ensure that analyses are based on all available data.</p> <p>Term of Reference d)  Indicators of state of sensitive species have been developed for fish in the North Sea. However, there are numerous ICES areas where a list of sensitive species is not available for fish. Even less information exists for other taxa. To allow advice on sensitive species throughout the ICES area, there is a need to expand the methods used to encompass other areas and to suggest a format in which risk based advice on sensitive species can be given.</p> <p>Term of Reference e)  To support the production of operational ecosystem advice, WGEKO will define a list of relevant pressure, driver and state indicators to be estimated by relevant experts groups, including stock assessment groups. This will provide guidance for groups which are currently interested in providing more ecosystem information for advice but unsure on exactly what to provide and in which format.</p>

Resource requirements	The research programmes which provide the main input to this group are already underway, and resources are already committed. The additional resource required to undertake additional activities in the framework of this group is negligible.
Participants	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities	None.
Financial	No financial implications.
Linkages to advisory committees	There are no current direct linkages with the advisory committees.
Linkages to other committees or groups	There is a very close working relationship with the groups of the Fisheries Technology Committee, WGBIRD, BEWG, WGBIODIV and WGSAM.
Linkages to other organizations	OSPAR, HELCOM

## Annex 4: Recommendations to other EGs

REQUEST	GROUP	EXPLANATION
Information on spatial distribution and abundance of sensitive benthos and benthic habitats	WKFBI, BEWG	In 2016, WGEKO has considered different measures of species and habitat sensitivity. WKFBI and BEWG is requested to consider this information in their work. Further, WGEKO requests that maps of sensitive habitats produced by WKFBI are made available to WGEKO for their analyses of the overlap of sensitive habitats with the distribution of sensitive fish species.
Annual spatial distribution of VMS from fishing vessels using towed gear	SGVMS	In 2017, WGEKO will continue the work on the effect of the landing obligation on benthic communities. As part of this, WGEKO will use spatial maps of the historic fishing intensity from VMS. For the areas where such data are not available online, WGEKO requests, where possible, similar data to those currently available for the North Sea.
Annual commercial catches in ton by ICES rectangle per species caught in demersal fisheries in the North Sea as far back in time as reliable data can be obtained	HAWG, WGNSK	In 2017, WGEKO will compare the historic distribution of sensitive fish and benthos species with that of fishing effort and catches to identify high-fishing-low-sensitivity areas and low-fishing-high-sensitivity areas.
Annual commercial catches in ton by per species and length group as far back in time as reliable data can be obtained	HAWG, WGNSK, WGWIDE	In 2017, WGEKO will compare the degree to which fisheries in different areas are balanced or specialised in specific species and length groups. For this effort, WGEKO needs catches by length group for the set of species accounting for 95% of landings in the following areas: Barents Sea, North Sea, Irish Sea, Celtic Sea, Bay of Biscay and Portugal.
Survey cpue per species and length group as far back in time as reliable data can be obtained	HAWG, WGNSK, WGWIDE	In 2017, WGEKO will compare the degree to which fisheries in different areas are balanced or specialised in specific species and length groups. For this effort, WGEKO needs Survey cpue per species and length group for the six areas listed above, if this is not available from DATRAS download, where possible corrected for differences in catchability.