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# Report of the Working Group on Multispecies Assessment Methods (WGSAM)

22–26 October 2012

Venice, Italy

ICES International Council for the Exploration of the Sea
CIEM Conseil International pour l'Exploration de la Mer

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## Contents

## Executive summary

This is the sixth report of the pan-regional Working Group on Multispecies Assessment Methods (WGSAM). The group met at the Ca' Foscari University of Venice in Venice, Italy, and reviewed ongoing multispecies and ecosystem modelling activities in each ICES ecoregion.

The participants provided an updated inventory, to supplement the information collated in 2007-2011 (ToR 'a' and 'b'). New information was presented for Barents Sea, North Sea, Kattegat/Skagerrak, Venice lagoon, Adriatic Sea, Baltic Sea and Northeast US. The group reviewed a new key run for the Baltic Sea SMS. A summary dataset of natural mortalities, stock numbers and biomasses of the modelled species was compiled and is available for download with the report.

WGSAM has continually worked towards significant development of new methods and improvements in model functionality. This year, WGSAM worked for the first time on evaluating and exploring the use of size spectra models (ToR 'f'), with a particular emphasis on investigating the effect of parameter selection on model predictions. Work on the development of cross-model validation techniques and suggestions of how to test various multispecies models using a common, virtual dataset were continued from last year and discussed along with the necessary characteristics of such datasets (ToR 'd'). The group further continued their work towards obtaining new stomach data from the ICES region (ToR 'c').

WGSAM, WGFE and WGECO all examined the development of foodweb and ecosystem indicators relevant to the marine strategy framework directive (MSFD descriptor 4) this year. To ensure knowledge transfer and coordination between the groups, WGFE and WGSAM met consecutively/concurrently and worked in a few joint session on this topic (ToR 'e'). Two joint members of WGECO and WGSAM attended the WGSAM/WGFE meeting. A range of indicators of foodweb and ecosystem Good Environmental Status were suggested, four of which (natural mortality by age, the proportion of total mortality which is caused by natural sources, the large fish indicator LFI, and the biomass in functional groups) were selected to present examples of multispecies advice.

The concept of Maximum Sustainable Yield and other biological reference points was explored in WGSAM using simulations based on stochastic age based models (Gadget, STOCOBAR and SMS) and fleet based biomass models (Ecopath with Ecosim), as well as with analytical considerations (ToR 'h'). The models showed that increasing predator stocks will result in decreasing yield of their prey. Balanced fishing was examined to determine if this harvest strategy could improve yield but this was not the case. For the North Sea, the rebuilding of the top predators cod and saithe resulted in increasing predation mortality of haddock and whiting, and these stocks could only be maintained within safe biological limits if fishing mortality was substantially higher than current single species estimates; again this should be received as a simulation result, not a recommendation. Even in this case, whiting was frequently at low biomass due to high predation mortalities induced by grey gurnard.

This year, WGSAM suggests a format for multispecies advice for inclusion in reports of other working groups and in ICES advice (ToR 'i'). The advice includes a general description of the most important species interactions, advice on community and foodweb indicators and advice on the combination of target Fishing mortalities pro-

ducing precautionary results or close-to-MSY in a multispecies environment. Examples are presented for the North Sea and Baltic Sea.

# 1 Opening of the meeting

The **Working Group on Multispecies Assessment Methods** [WGSAM] met at University Ca'Foscari in Venice, Italy from 22–26 October 2012. The list of participants and contact details are given in Annex 1. One of the two Co-Chairs, Anna Rindorf (DTU-AQUA, Denmark) welcomed the participants and highlighted that like last year, the Working Group had a broad geographic scope, this year encompassing research in previous WGSAM meetings but particularly including Mediterranean ecosystems. The Terms of Reference for the meeting (see Section 2) were discussed, and a plan of action was adopted with individuals providing presentations on particular issues and allocated separate tasks to begin work on all ToRs.

## Acknowledgements

WGSAM would like to thank Fabio Pranovi for logistics during the meeting and Claire Welling of the ICES Secretariat for her continued support with the WGSAM SharePoint site.

# 2 Terms of reference

The **Working Group on Multispecies Assessment Methods** (WGSAM) chaired by Anna Rindorf, Denmark and Jason Link, US, will meet in Venice, Italy, 22–26 October 2012 to:

a ) Review **further progress** and report on key updates in multispecies and ecosystem modelling throughout the ICES region;

b ) Report on the development of **key-runs** (standardized model runs updated with recent data, and agreed upon by WGSAM participants) of multispecies and eco-system models for different ICES regions (including the Baltic Sea, and others as appropriate);

  i ) If needed, run additional model runs to those made by WGSAM in 2012 for the North Sea in order for fulfilling ToR b)

  ii ) Develop a draft advice text for the ICES Advice Report 2013 for a multispecies management plan for the North Sea along the lines done in 2012 for the Baltic.

c ) Work towards implementing new **stomach sampling** programmes in the ICES area in the near future;

d ) Explore how '**virtual multispecies datasets**' (including survey, catch and stomach content data) for use in multiple multispecies models, especially for comparison and sensitivity testing, could be constructed;

e ) Develop **foodweb and ecosystem indicators (descriptor 4)** relevant to the marine strategy framework directive (MSFD) from outputs of model key runs and input data;

f ) Evaluate and explore **size spectra models** and compare to multispecies models in use in the ICES region;

g ) Explore the **trophic role** of pelagic cephalopods (i.e. squid);

h ) Explore the concept of Maximum Sustainable Yield (**MSY**) within a multispecies context and how it affects other biological reference points (BRP);

i ) Based on various assumptions about policy choices, suggest a format for multispecies advice for single species groups and present examples of such advice for areas where the necessary information is available.

iii ) As a minor part of the work consider whether the WKBALT work needs further multispecies considerations for being ready as a basis for the advice for Baltic stocks in 2013, and if so, suggest a draft advice text for the ICES Advice Report 2013 for the Baltic Sea.

*Of these, a, b and i are standing terms of reference, while c and d are 'multiyear projects'*

*Longer term aspirations (possible ToRs for future years)*

Review estimates of abundance and productivity at **lower trophic levels**, and work towards the inclusion of such information in multispecies models.

Evaluate the major **sources of uncertainty** when making projections using multispecies and ecosystem models and explore possible best practices for addressing said uncertainties.

Explore the **trophic role** of other non-assessed, but suspected important predator species (e.g. gurnard, starry ray).

# 3 ToR a): Review further progress in multispecies and ecosystem modelling throughout the ICES region

## 3.1 Ecoregion A: Greenland and Iceland Seas

Work is ongoing in Iceland investigating how marine mammals can be included in Gadget, with a model of their population dynamics examining how data on marine mammal diet and abundance can be included. The Gadget model is used in a number of single species assessments and extended single species assessment of shrimp where the natural mortality is linked to the abundance of cod in the shrimp survey. Related to these single species models, work is ongoing to estimate "appropriate" weights on different likelihood components in Gadget, ultimately to obtain more refined confidence intervals by bootstrapping the data. This work could be extended to a multispecies Gadget model for Icelandic waters.

## 3.2 Ecoregion B: Barents Sea

In the Barents Sea, cod abundance and spatial distribution has both increased strongly in recent years (ICES CM 2012/ACOM:05), and cod spawning stock size is at the highest level recorded in the time-series going back to 1946.Thus, it is important to include effects of food abundance on cod growth and maturation in multispecies models. Also, the population dynamics of large cod, including cod cannibalism (Yaragina *et al*., 2009) needs to be modelled in an appropriate way.

### 3.2.1 Gadget models

The SYMBIOSES project to produce a linked series of models for exploring risk relating to oil development is continuing. The larval model has now been linked to plankton and oceanography model, and work to link the fish population model will take place in 2013.

Recent years have seen summer sea ice cover in the Barents Sea reduced. As a result species such as cod, which were previously confined to the southern Barents Sea,

have been increasing numerically in the northern and eastern Barents Sea. The Gadget model has therefore been extended to include a spatial component in the Barents Sea to be able to examine the likely consequences of this trend. The Gadget model is also being used to examine what would be the consequences of "balanced" fishing in the Barents Sea.

### 3.2.2 ATLANTIS

Work is progressing to develop an ATLANTIS multispecies model for the Barents and Nordic (i.e. Norwegian) Seas. This model covers the major benthic, planktonic, fish, seabird and marine mammal components of the Barents Sea ecosystem, as well has having a detailed three-dimensional spatial structure and oceanographic inputs from the ROMS oceanography model. A first version of this model is now running and is undergoing testing and further parameterization.

### 3.2.3 STOCOBAR

During the last year further development of the STOCOBAR model involved modelling additional ecosystem components. It was designed to transform the multispecies STOCOBAR model into the ecosystem EFIBAR model. EFIBAR (**E**cosystem and **FI**sheries in the **BAR**ents Sea) is designed for study of all ecosystem components, however the commercial fish species will remain the priority. It may be considered as an ecosystem-based extension of the STOCOBAR model.

The general scheme of the EFIBAR model is presented in the Figure 3.2.1. The number of additional ecosystem scenarios is unlimited in the model and is determined by availability of input data. In the present model version the additional ecosystem scenarios depend on simulated water temperature, cod and capelin stock parameters. However, feedback through the simulation of cod recruitment, growth, feeding, maturation and natural mortality is not yet implemented.

The main principles of producing ecosystem scenarios in the model are:

1 ) Ecosystem scenarios are based on stochastic distributions and statistical relations of ecosystem components, which are derived from historical data or input as model assumptions.

2 ) Projections of additional ecosystem components are modelled through the implementation of single stochastic relationships of these components coupled with available model outputs.

3 ) Simulated time-series of ecosystem parameters integrates the contributions from the different single stochastic relationships by using their weighting factors.

The developing method allows us to create in the STOCOBAR model the ecosystem scenarios, which are internally consistent and biologically realistic. They are may be applied for testing ecosystem consequences of different climate change scenarios in the Barents Sea as well as for the evaluation of impacts on the ecosystem of different strategies of cod fisheries.

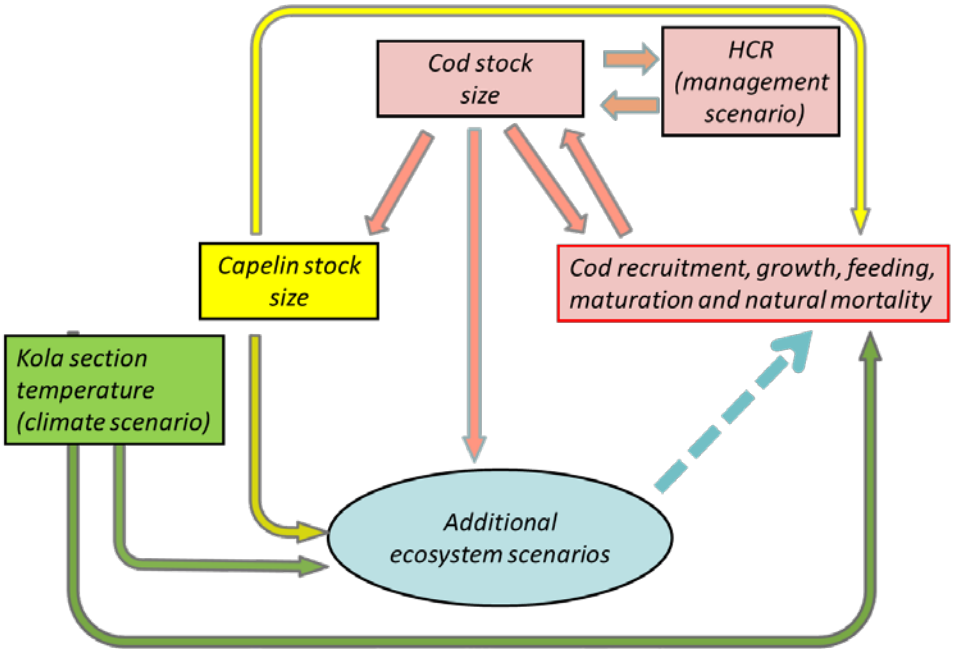


**Figure 3.2.1. EFIBAR model structure.**

## 3.3 Ecoregion C: Faroes

There is no progress to report on multispecies modelling in Ecoregion C this year. There were no participants present at the 2012 meeting from this Ecoregion.

## 3.4 Ecoregion D: Norwegian Sea

The Atlantis model (described in 3.2.2) covers both the Barents and Nordic Seas. The Gadget model (see 3.2.1) is focused on the Barents Sea, but also includes herring in the Norwegian Sea. In models for this area, it is important to take into account the increase northwards in the distribution area of Northeast Atlantic mackerel in recent years.

## 3.5 Ecoregion E: Celtic Seas

### 3.5.1 Ecopath in the Celtic Sea

Work on modelling the Celtic Sea ecosystem and effects of fisheries and climate on Seabirds has been published in Dr Valentina Lauria's PhD thesis, with plans for work to be published in the primary literature. Cefas is working toward developing a spatial 'Ecospace' model of the Celtic Sea ecosystem. A GIS database has been completed with all environmental, ecological and fishery data layers that will be used in developing the spatial model.

## 3.6 Ecoregion F: North Sea

### 3.6.1 Ecopath with Ecosim in the North Sea

Increasing demand and expectations for ecosystem models to become useful operational tools is influencing the direction of developments in this area. Three particular areas of development on EwE modelling are (i) representation of fleet structure/ segmentation, (ii) validating modelled spatial distribution of fishing, (iii) evaluating impacts of uncertainty in model parameters.

*(i) Representation of fleet structure/ segmentation*

Work on how best to represent fishing fleets in ecosystem models is premised by the need for modelling tools capable of evaluating the relative impacts of different fleet segments under a variety of alternative management scenarios. A conceptual approach currently being pursued at Cefas shifts the focus of attention of ecosystem models away from the details and complexity of the foodweb interactions to a more explicit consideration of the fleet-web (Figure 3.6.1). Experience working with stakeholders shows that this representation of how fishing interacts with the ecosystem is much more intuitive and leads to better engagement on collaborative work on long-term management plans (e.g. EU FP7 – GAP2 and MYFISH project work). Because EwE model simulations assume that the behaviour of fishing fleets does not change (i.e. their targeting of species is fixed), it is important that the model should aggregate fleets at a level where this assumption is considered reasonable. DCR and DCF data are being used understand how stable the catch compositions are at different levels of fleet aggregation (Figure 3.6.2).

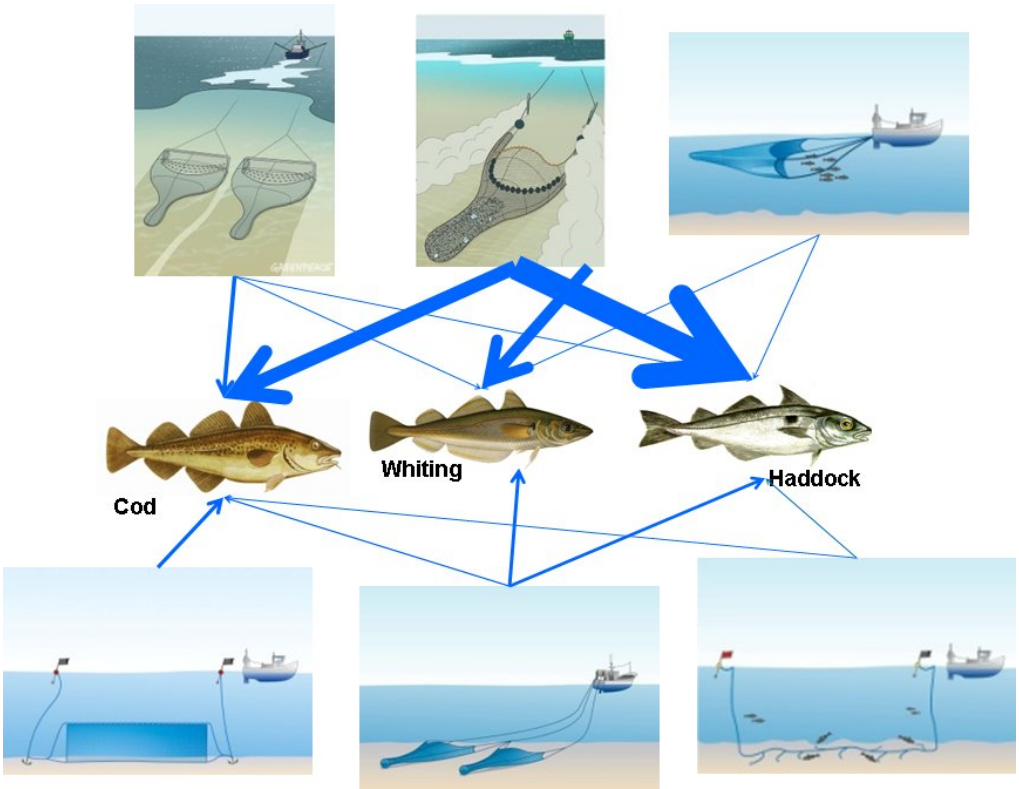


**Figure 3.6.1. The 'fleet-web' (Mackinson pers. comm.). Focusing on the interaction of fleets and species. Thickness of arrows represents the fishing mortality of each fleet one each species.**

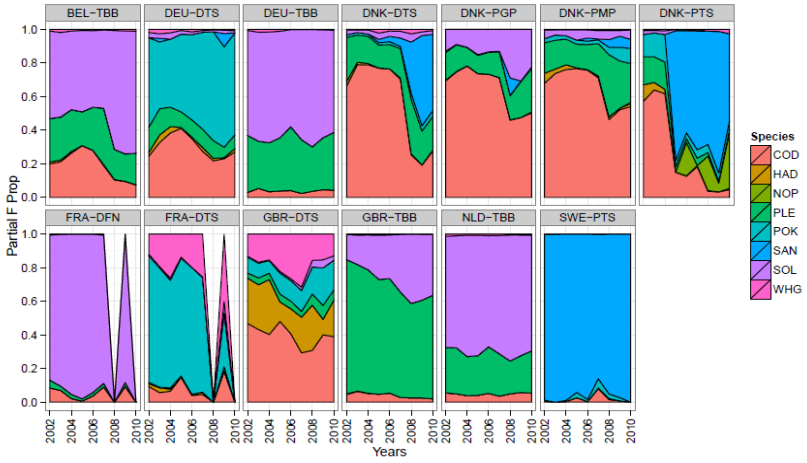


**Figure 3.6.2. Changes in the proportion of mortality that each fleet exerts on each species.**

*(ii) Validating modelled spatial distribution of fishing*

Building on the North Sea EwE model Key Run published in ICES WGSAM 2012, work is in progress to make a spatial version of the key run model. The work being undertaken by collaboration between Cefas and the University of Oslo is using the previously published version of the spatial model (Mackinson and Daskalov, 2007) to refine parameters, test the methods for assigning functional groups to habitats and evaluate the model predictions of species and fleet distributions.

*(iii) Evaluating impacts of uncertainty in model parameters.*

Being explicit about how parameter uncertainty influences model predictions of alternative harvest strategies is important because it affects how predictions are interpreted and what decisions might be made. Cefas is developing a routine for sampling Ecopath and Ecosim parameters and evaluating their impact on the outcomes of alternative management scenarios. The EwE plug-in is intended to (1) sample parameters from user specified distributions, (2) set harvest control rules and (3) evaluate the HCRs performance by predicting (i) the biomass of functional groups, (ii) the catch by fleets, (iii) indicators (to be defined) and estimating their associated uncertainty.

Contributing to the EU FP7 MEECE project, a coupled physical-biogeochemical-foodweb model (Ecosim linked to GOTM-ERSEM, Beecham *et al*. (in review)) has been used to explore 10 different scenarios reflecting different options for future patterns of fishing, climate and eutrophication (Figure 3.6.3). The work shows that, in general though there are some broad conclusions about some of the trends that might be experienced as a result of changes in production, the highest trophic level species respond positively to less fishing and more nutrients, whereas the effects on demersal and flatfish are smaller. Moving to an MSY-based fishing approach clearly benefits the fish that are fished less whereas their competitors may be adversely affected.

Access to selected model results on how changes in a variety of human and environmental pressure may affect North Atlantic waters is ensured via the MEECE Atlas. Complete access to hindcast model simulations and selected results from the projections will be ensured via ICES WGOOFE working group and website (www.wgoofe.org/ICES).

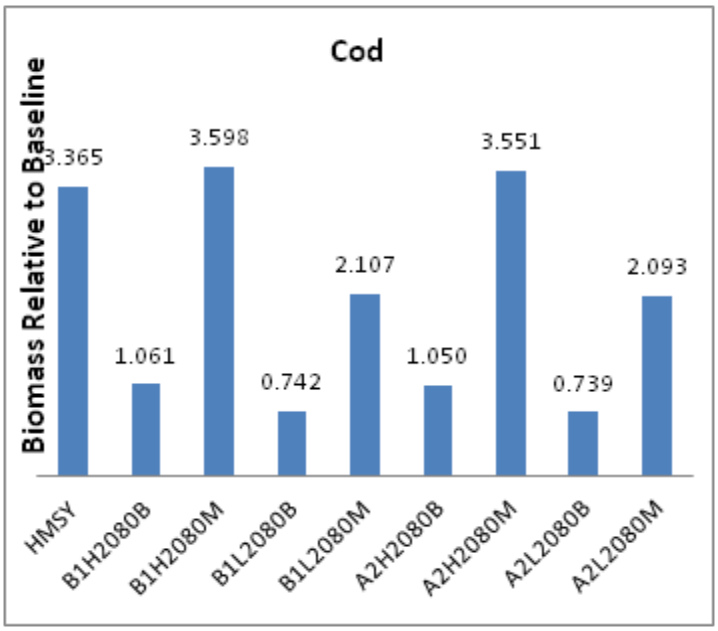


**Figure 3.6.3. Effect of interaction of climate change, fishing and nutrient reduction scenarios on cod biomass in the North Sea relative to current baseline levels.**

Additional work on how fisheries and environment influence ecosystem the dynamics of the North Sea ecosystem are in preparation for publication entitled 'Empirical and model-based evidence explain changes in the North Sea ecosystem linked to environmental change and fishing' (Mackinson in prep).

### 3.6.2 Ecopath with Ecosim in the Eastern Channel

In a collaboration between Cefas and the University of Kent, UK, a spatial EwE model of the Eastern Channel is being used evaluate the ecosystem and fishery effects of the size and location of marine protected areas derived from Marxan analysis. Doing so requires ensuring that the distribution of fishing fleets modelled in EwE matches closely with the distribution of fishing activity data (based on VMS) used as a 'cost' layer in Marxan. The approach being taken to this is a pragmatic one: first the catch composition of fleets in EwE are updated to match the year of the VMS, then the VMS data used in the Marxan analysis is used in Ecospace, together with specification of port locations, to influence the Ecospace prediction of the distribution of fishing fleets.

### 3.6.3 Ecopath with Ecosim for the southern part of the North Sea

Parameterization of an EwE model for the southern part of the North Sea has been initiated at the Thünen Institute in Hamburg. The aim is to explore the consequences of different MSY derivatives in a multi species and mixed fisheries context. The model will be based on the Cefas North Sea EwE model but will be restricted to ICES areas IVb&c. It will focus on flatfish and brown shrimp and represent a number of non-target species as well as marine mammals and seabirds. The fishery is represented by the main fleets fishing in the southern part of the North Sea. First Ecosim runs utilizing the policy search tool to maximize yields under different weightings of conservation, economic and social constraints are foreseen during spring 2013.

### 3.6.4 Linking SMS to the output of the NPZD model EOCHAM-4 and species distribution models

The lower trophic level model ECOHAM-4 has been used to identify the most important drivers of fish recruitment per spawning-stock biomass. In 0.5 x 1 ° North Sea rectangles, passive drifters were started each day during the period 1980 to 2006 and measured parameters such as temperature, salinity, phytoplankton and zooplankton concentrations. Further, drift parameters like direction and distance were monitored. Thus for each starting rectangle a time-series of average conditions experienced by the drifters was generated. These time-series were split and the period 1980-1996 was used to identify significant correlations between R/SSB or zooplankton and the experienced Proxy series. From those correlations that still hold in the second half of the time series (1997–2006) the best (highest $r^2$ in the resulting first and second correlation matrix) 15 proxies were chosen. For the forecast (2071–2099) the ECOHAM-4 was forced with a downscaled climate model projection and used the fore determined proxies to project R/SSB, phytoplankton and zooplankton abundance under this climate change scenario. The predicted changes in recruitment success (decreasing: Herring, Norway Pout, Whiting; increasing: cod, sandeel, sole, sprat) were transferred to the stochastic multispecies model (SMS) by modifying the stock recruitment relationships fitted in SMS hindcasts. By applying currently utilized $F_{msy}$ proxies from single species assessments, species-specific stock sizes and yields were projected into the future. Interestingly and despite a slightly increased R/SSB trend for cod, the biomass of cod decreased in the modified projection (with the changed recruitment dynamic) in contrast to the normal (standard stock recruitment dynamics) SMS projections, mainly because alternative prey as Herring, Whiting, Haddock, and Norway Pout decreased and predation pressure on cod recruitment increased. Only species in the bottleneck or lower trophic positions like sprat, sandeel or flatfish like sole and plaice increased in abundance.

In the EU project VECTORS, plans to link the output of different species distribution models to SMS are extant, primarily by modifying future predator–prey overlap based on the predictions of stock distributions under climate change scenarios. Due to the described linkages, the model framework can be used to evaluate climate change scenarios and consequences for future fisheries yield can be predicted.

### 3.6.5 ATLANTIS model

Work on implementing Atlantis in the North Sea has just begun in the FP7 project VECTORS to analyse potential impacts of management measures on the ecosystem and economy. Specifically, the effects of installing wind parks in the North Sea will be investigated, as well as various fishery closures and marine protected areas. The focus has been on the basic settings of the simulated areas (polygons) and basic parameterization will hopefully be finished in summer 2013.

### 3.6.6 Kattegat/Skagerrak

At present there is a lack of information available about the historical development of fisheries in this area relative to changes in foodweb structure and function. A recent study presented at the WG on Mean Trophic Level (MTL) and Primary Production Required (PPR) for the Swedish fisheries in the area indicates that there is a need for further development of applicable foodweb indicators. Synergetic effects from historical overfishing and coastal eutrophication are at present difficult to disentangle (Baden *et al.*, 2012), and testing suitable foodweb indicators is important to allow

monitoring of GES in relation to MSFD requirements for the area, as well as for fisheries advice in a multispecies context.

## 3.7 Ecoregion G: South European Atlantic Shelf

There is no progress to report on multispecies modelling in Ecoregion G this year. There were no participants present at the 2012 meeting from this Ecoregion.

## 3.8 Ecoregion H: Western Mediterranean Sea

There is no progress to report on multispecies modelling in Ecoregion I this year. There were no participants present at the 2012 meeting from this Ecoregion.

## 3.9 Ecoregion I: Adriatic–Ionian Seas

### 3.9.1 Foodweb model applications in Mediterranean lagoons

The steady-state foodweb model (FWM) of the Venice lagoon was updated by using a dataset of biomass density and landings collected during the 2003–2005 time frame. The underdetermined system of mass-balance equations was solved by means of an Inverse Methodology of linear estimation, and results were compared with previous Ecopath models, focused in the 1990s, in order to detect trends in ecosystem evolution (Brigolin *et al.*, 2011). The analysis indicated that the main patterns on energy flow are consistent with previous estimates. With respect to the role of detritus and benthic-pelagic energy exchanges (see Figure 3.9.1); results indicated that during the last decade the Lagoon ecosystem was accumulating energy in the sediment, although detritus recycling by the ecosystem was slower compared to the 1990s.

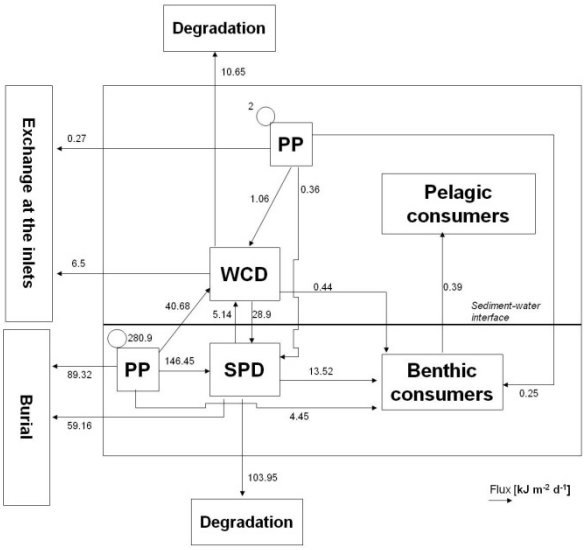


**Figure 3.9.1. Most important energy fluxes (kJ $m^{-2}$ $d^{-1}$) between benthic and pelagic systems. (from Brigolin *et al.*, 2011).**

The role of space was investigated by trying to relate explicitly the functioning of the ecosystem with its spatial configuration in habitats. In this exercise the temporal dimension was considered only at the ecosystem scale, i.e. as changes in spatial configuration, and assuming that habitat structure and functioning can be treated at steady-state. At the ecosystem scale, indices of functioning were calculated on a weighted set of energy fluxes, calculated by taking into account the relative extension of the differ-

ent habitats. The method was tested by comparing long-term and short-term scenarios induced by changes in the bathymetry, and short-term changes in the area in which Manila clam fishery is currently allowed.

Influence of metabolic rates variability on FWM output was investigated by estimating the specific production P/B of single-species compartments: this was carried out by means of a dynamic individual bioenergetic model. Main objectives of these runs were:

- to assess the effect seasonal fluctuations of water temperature on the estimation of energy and matter flows across a foodweb;
- to test the sensitivity of foodweb model results to inter-annual variability of water temperature and size-structure of the population.

Results (see Brigolin and Pastres, 2012) indicated that in such a temperate environment, the metabolic rate estimates obtained from an average water temperature and body size may lead to energy fluxes which are not representative for the whole time frame of a steady-state model. The combined use of dynamic growth models and foodweb models provided a way to test the sensitivity of the result of the foodweb model to interannual fluctuations of water temperature and population age/size structure.

### 3.9.2 End-to-End modelling for the Adriatic Sea

The Adriatic Sea is a well-studied area with biogeochemical/physical (BGC; Cossarini and Solidoro, 2008; Querin *et al*., 2006, Lazzari *et al*., 2012) and foodweb (Coll *et al*., 2007; Pranovi and Link, 2009) models. Integration of physical-biogeochemical and foodweb models allows for using extant and well established tools for representing both fisheries and climatic effects on foodwebs.

Among the many possible climatic effects, we studied those induced by changes in precipitation patterns. In previous downscaling experiment for the north Adriatic, meteorological data were extracted from the output of three multidecadal RegCM simulations (Gao *et al*., 2006), one for the present-day period of 1961-1990 (hereafter referred to as RF experiment), and two future scenario simulations for the period 2071-2100 under the A2 and B2 emission scenarios of IPCC (2000), in which the CO2 concentration increases, respectively, to about 850 ppm and 600 ppm by 2100, respectively. Results in terms of spatio-temporal dynamics of biogeochemical properties provide evidence of significant impacts of climate change: under both the A2 and B2 scenarios we observe an amplification of the seasonal precipitation patterns, with drier summers and wetter winters, which affect the timing of nutrient inputs to the lagoon (Cossarini *et al*., 2008). The insights gained from these previous experiments for the North Adriatic area were used to build plausible scenarios of changes in timing and quantity of nutrient input to the system. Therefore three scenarios have been implemented, i.e. a reference scenario and two climatic scenarios (A2 and B2), concerning different nutrient input as inorganic phosphate during the year. These input scenarios have been applied to the biogeochemical model first and then have been applied by means of the E2E approach.

The biogeochemical and foodweb model are integrated through a two steps procedure (Libralato and Solidoro, 2009). In the first step the models are integrated by extending the EwE model for including the main biogeochemical processes, thus accounting for all possible interactions among upper and lower trophic levels. In the second step the nutrient inputs are adjusted in the extended model to adjust to the

biogeochemical results in terms of nutrient dynamics. The adjustments are necessary for accounting differences in parameterization of the two models, including differences in the time and space scheme used.

An Ecopath with Ecosim (EwE) model representing the foodweb of the North-Central Adriatic Sea (Coll *et al*., 2007) is forced by a biogeochemical (BGC; Cossarini *et al*., 2008) model of the system which, in turn, simulates two future scenarios, one with changes in the seasonality of nutrient inputs (equivalent to A2) and another that also include a general reduction of inputs of nutrient from the rivers due to the implementation of sustainability policies in the land use management (reduction of fertilizers; B2).

E2E scenarios consisted of 30 year simulations. In future climatic scenarios the climatologic conditions were kept for the first two years and then the new input conditions were changed for the next 28 years, according to the average seasonality obtained from the biogeochemical outputs scenarios. For each of the three climatic scenarios fisheries scenarios were representing by changing fishing mortality (F +/- 25% and +/- 50%) for the main fished species, i.e. anchovy, sardine, hake and red mullet. Moreover additional scenarios were applied to represent changes in effort by fishing fleets (E +/- 25%) for midwater trawl, beam trawl, bottom trawl, purse-seine and tuna fishing. A total of 81 scenarios (climate+fishing) were analysed by comparing the long-term evolution of biomassand applying a set of group-based and ecosystem based indicators. These include foodweb biodiversity indicators (e.g. Kempton index) and complexity indicators (e.g. System Omnivory Index). The results are synthesized by highlighting antagonistic and synergistic effects between fishery and climate, showing that both future climate scenarios effects are counteracted by reduction of fishing pressure. This end-to-end work is going to be further developed in FP7 projects OPEC and PERSEUS.

## 3.10 Ecoregion J: Aegean–Levantine

There is no progress to report on multispecies modelling in Ecoregion J this year. There were no participants present at the 2012 meeting from this Ecoregion.

## 3.11 Ecoregion K: Oceanic Northeast Atlantic

There is no progress to report on multispecies modelling in Ecoregion K this year. There were no participants present at the 2012 meeting from this Ecoregion.

## 3.12 Ecoregion L: Baltic Sea

ICES, with several institutes around the Baltic, has for decades invested substantially in the research on multispecies interactions, ecosystem functioning and integrated assessment. Currently, there exist several multispecies and ecosystem models for the Baltic Sea (for an overview cf. ICES, 2012a). SMS results were scrutinised in more detail as they were considered for use in management advice (ICES, 2012b, STECF, 2012).

The results presented in this section of the report are a partial summary of the FACTS project work in STECF and WKMULTBAL. Much greater detail is available in the reports on these two expert groups.

There are some concerns regarding the capability to model multispecies aspects predictably for the future. In particular the multispecies aspects depend on predation data mainly from the 1980s and there is an urgent need to update the information

base (see ToR c). Furthermore, the current regime – in terms of productivity and spatial distribution of fish stocks in the Baltic – is different from the earlier period when diet data were collected.

Management of fisheries for cod can have an impact on fishing opportunities for sprat and herring, and vice versa. The main prey of cod is sprat and, to a lesser extent, herring and juvenile cod (cannibalism). Additionally, herring and sprat sometimes feed on the eggs of cod. Furthermore, growth of herring and sprat has been density-dependent (Figure 3.12.1), and growth of cod has, to some extent, been dependent on herring and sprat biomass.

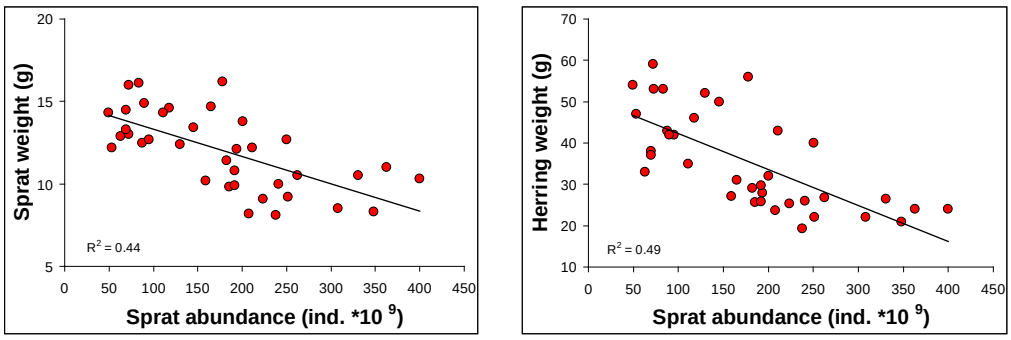


**Figure 3.12.1. Relation between clupeids mean weights at age 3 and sprat total abundance (data from WGBFAS 2011).**

Finally, the relative distributions of predator (cod) and prey (herring and sprat, possibly juvenile cod) have changed substantially during the last years, and for the time being much herring and sprat are outside the predatory reach of cod (Figures 3.12.2 and 3.12.3).

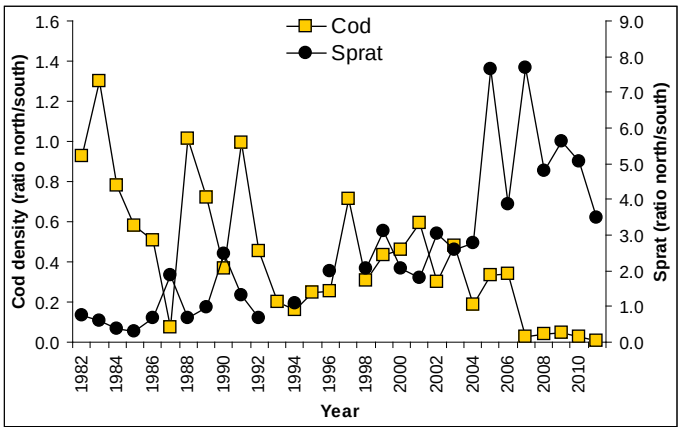


**Figure 3.12.2. Changes in spatial distribution of Eastern Baltic cod and sprat during the past 30 years. This is indicated as density ratio between northern areas (SDs 27–29) and southern areas (SDs 25–26) from acoustic (sprat) and bottom-trawl surveys (cod). Modified from Casini *et al.* (2011).**

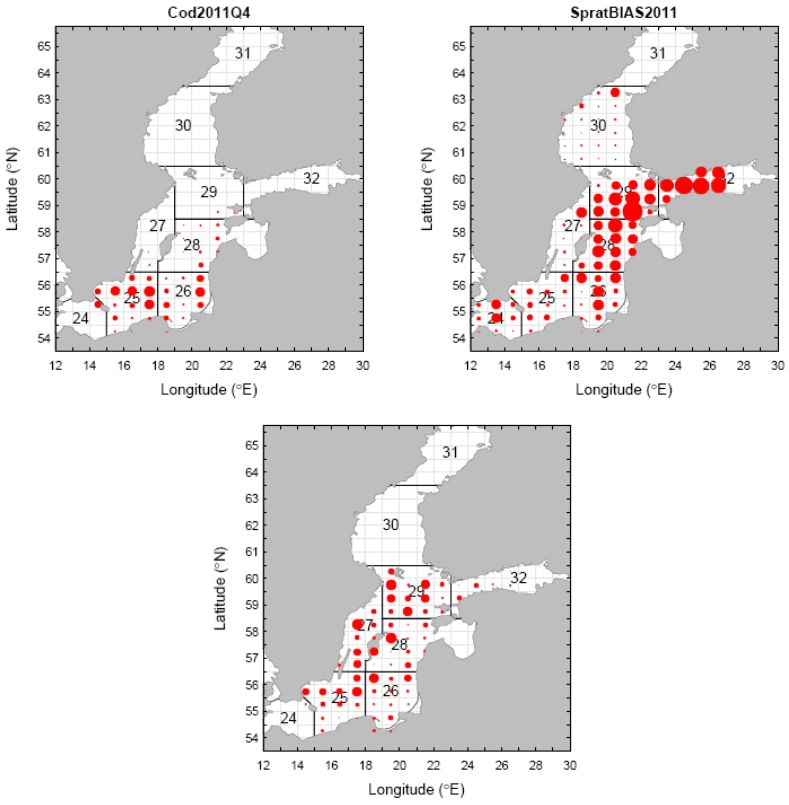


**Figure 3.12.3. Spatial distribution of Eastern cod, sprat and Central herring in 4th Quarter 2011, from acoustic survey (BIAS, sprat and herring) and bottom-trawl survey (BITS, cod).**

The combination of an increasing cod stock and low abundance of sprat and herring in SD 25 (in the main distribution area of cod) has resulted in the lowest biomass of clupeids per cod currently available in this area since the 1970s. In line with low biomass of clupeids in the area, the mean weight of older cod (age-groups 4-7) in SD 25 has sharply declined since 2007 (Figure 3.12.4).

The mean annual growth rate of grey seals in the Baltic has averaged 7.5% annually during the last decade. In 2010, a total of approximately 23 100 grey seals were counted. The increase in stock size was highest in the northern areas and the predation pressure of grey seals on clupeoids has increased accordingly. The diet of grey seal in the Baltic consists of approximately 20 fish species. The most abundant prey items in the Baltic proper are Baltic herring, sprat, and cod, and in the Bothnian Sea and Bothnian Bay Baltic herring, *Coregonus* sp., Baltic salmon, and sea trout. An adult seal consumes on average 4.5 kg fish per day, of which 55% is clupeoid prey in the Baltic Main basin and 70% in the Bothnian Sea and Bothnian Bay. According to acoustic estimates, predator– prey distribution patterns, migration patterns, and multispecies analysis (SMS), the predation effect of grey seals on Baltic herring and sprat stocks is still at a very low level. Hence, with present grey seal stock sizes, the impact of seal predation can be ignored in whole Baltic-scale herring and sprat stock management considerations. Locally, however, grey seal–fishery interactions play an important role and should be taken into account in future spatial planning and ecosystem management.

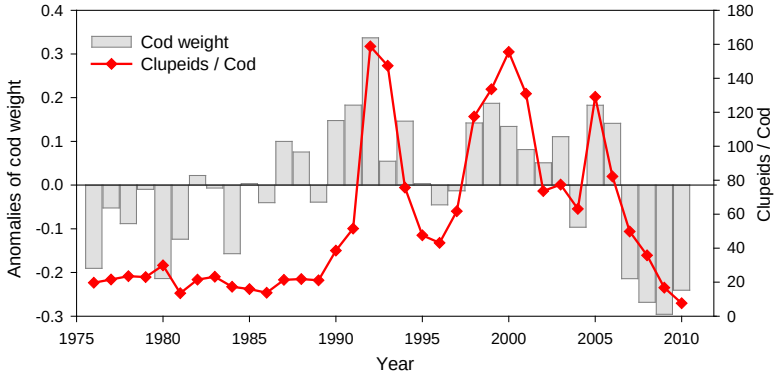


**Figure 3.12.4. Anomalies in mean weight of cod (average of age-groups 4–7) in SD 25 (bars) compared to changes in the biomass of clupeids (sprat and herring) relative to the number of adult cod (at age 4 and older) in the same area (line; Eero *et al.* submitted).**

While the effects of spatial distributions of predator and prey can be assessed in the retrospective runs of the SMS model, for forecasting there is limited knowledge of the processes that lead to changes in spatial distributions. Moreover, when taking clupeid density-dependent growth in consideration, the $F_{msy}$'s estimated by SMS are very high for both herring and sprat, and the reason for this should be further investigated. For these reasons, both STECF (2012) and ICES (2012b) decided to use the 'one-area' option and also ignored density-dependent growth in Management Strategy Evaluations (MSE). However, the sensitivity of this model configuration to including area-dependent predation mortalities and density-dependent growth has been tested in FACTS via applying observed status quo conditions in SMS forecasts.

All the multispecies $F_{msy}$ values for Eastern Baltic cod, Central Baltic herring and Baltic sprat are higher than the single species values. Particularly for cod and sprat higher Fs give very similar yields for the long term and will give lower SSBs and in some cases risks of stock decline to the "lower biomass" reference points (that is a first suggestion for a lower SSB to avoid impaired recruitment). Model results indicate that although higher Fs on Eastern Baltic cod give little increase in cod yield, a higher cod F gives higher yields from Baltic sprat and Central Baltic herring. As current modelling for $F_{msy}$ does not include any structural uncertainties, risks of stock decline and impaired cod recruitment will be higher than those estimated. The presence of yearly constraints for changes in cod TAC increases the variability of stock size, with the increases greater in a multispecies system (for detailed results, please cf. STECF 2012; in this report only the main results will be presented).

The present distribution pattern, with a limited distribution range for cod (concentrated in the southern area) and basin wide distribution for herring and sprat (but mainly concentrated in the northern areas, at least in some seasons; Figures 3.12.3 and 3.12.4), imply that an increase in F on cod, will not necessarily result in increasing Baltic-wide clupeid stock sizes. Conversely a decrease in F on cod will not necessarily result in a decrease of the Baltic clupeid stock size if it will not be accompanied by a cod expansion into northern areas. However, cod cannibalism will be higher and limited growth of cod due to food deprivation will become a bigger problem. On the other hand, a reduction of clupeid F in Subdivision 25 will likely improve growth and condition of cod as well as reduce cannibalism. An increase in clupeid F in northern

areas (SDs 27–32) will likely not have a negative effect on cod, since this will not affect the stock component distributed in southern areas (SD 25-26). Further, a higher F on clupeids in northern areas would likely reduce density-dependence and improve the growth and condition of clupeid stocks (ICES, 2012b).

Higher $F_{msy}$ proxies for herring and sprat are also obtained when density-dependent growth is assumed for the two species, as the stocks compensate by a higher growth at lower stock densities due to either higher fishing mortalities or predation.

STECF (2012) concluded that 'If management is to follow the higher Fs associated with multispecies interactions it is very strongly recommended that biological sampling to support the interspecific relationships be restarted.'

### 3.12.1 Structural uncertainties in ecosystem modelling

#### Cod cannibalism

Cod cannibalism in the Eastern Baltic Sea has been estimated to be most intense in 1978-1984, a period with high juvenile abundance and large adult stock size. Multispecies modelling predicted about 60% of the 0-group and 30% of the 1-group cod were consumed by adult cod (Neuenfeldt and Köster 2000). Subsequently, cod recruitment and adults decreased whereas sprat, the main fish prey for cod, became significantly more abundant. Estimated predation rates on 0-group cod in the fourth quarter of the year and annually on 1-group cod decreased to 23% and 9%, respectively. No stomach data has been collected since 1994, however, model predictions predict that since 2003, when cod stock size has increased again, predation on the 0-group increased, too. However, predation on the 1-group has increased more slowly, because to consume 1-group cod, the individual adult cod have first to grow larger first.

Currently, the cod spawning period is delayed (from mainly spring to mainly summer) as compared to the late 1970s early 1980s. Therefore, the overlap between juvenile cod and adult cod might have changed. However, no stomach samples are available since 1994. Hence it is important to collect new stomachs of large cod to update the database on cod cannibalism. Therefore, given the limited number of stomachs sampled the rates computed have a large uncertainty for estimated cannibalism rates

#### Cod growth

Cod growth is not, yet, included in the model. For a review on problems in modelling cod growth, see STECF (2012). In addition to somatic growth, the individual food consumption by size has not been modelled either.

#### Clupeid growth

Clupeid somatic growth has been implemented as purely density-dependent. The rational for this is that when clupeid abundance/biomass increases, the individual growth of sprat and herring slow down, likely because of food competition (ICES, 2012b).

However, ICES (2011) and STECF (2012) concluded that more work is needed to fully understand the results of the runs in which density-dependent growth was included. In the actual runs, the total clupeid stock size was used as density-dependent factor. The current literature however shows that sprat is able to control the common food

resources and therefore drive the density-dependence, less so herring. Therefore, an enhancement of the model could be constituted by the inclusion of a sprat-only density-dependent factor, affecting both sprat and herring growth. Several investigations have pointed out that the clupeid growth is mainly density-dependent (through intra- and interspecific competition), but the inclusion of hydrological drivers in the growth models input for the SMS runs might provide an additional improvement.

#### Predation on cod eggs and competition for food between cod larvae and sprat

The change in the timing of spawning might have consequences for predation on cod eggs and larvae, and food competition between cod larvae and mainly sprat might also have changed. Both of the issues necessitate a new sampling program for cod, herring and sprat stomachs. The latest data are from 1994. Currently, predation on cod eggs is not included in the model.

#### Spatial considerations

Currently, it is assumed in the SMS model that the overlap between cod and clupeids is taken as the mean for the whole area, and that the overlap remains unchanged during the model period. Given the current spatial distribution of the stocks, the overlap between cod and clupeids are to a large extent limited to SD 25; there is less overlap in SD 26 and almost no overlap between cod and clupeids in the northeastern areas (SD 28–32). In SD 25, there are indications that cod is suffering from food limitation, with high predation pressure on sprat and herring in this area. On a Baltic wide scale, the impact of cod predation on sprat is limited, as the highest densities of sprat occur in northeastern areas, where cod is rare. Similarly for herring, the overall predation mortality in SD 25–32 is estimated relatively low. In northeastern areas, the density-dependent processes on sprat and herring growth are more pronounced, related to high densities of clupeids in these areas.

The reason for not integrating the spatial considerations in the 'best possible' forecast runs was that too little is known about mechanizm leading to future changes in the species' distributions. However, due to the changes in distribution of the three stocks in the Baltic, estimation of the trade-offs between species at different exploitation rates should take the overlap between the species into account (ICES, 2012b).

## 3.13 Ecoregion M: Black Sea

There is no progress to report on multispecies modelling in Ecoregion M this year. There were no participants present at the 2012 meeting from this Ecoregion.

## 3.14 Ecoregion: Canadian Northwest Atlantic

There is no progress to report on multispecies modelling in Ecoregion N this year. There were no participants present at the 2012 meeting from this Ecoregion.

## 3.15 Ecoregion: US Northwest Atlantic

### 3.15.1 Surplus Production Modelling

Two workshops (funded by the Comparative Analysis of Marine Ecosystems, CAMEO) were held in Woods Hole, MA with the goal of exploring the drivers that delineate the productivity of marine ecosystems, using surplus production models as a unifying analysis tool. A vast literature identifies three main processes that regulate the production dynamics of fisheries. These are collectively referred to as the triad of

drivers, and include: biophysical, exploitative, and trophodynamic processes (Figure 3.15.1). There were 13 northern hemisphere ecosystems from Canada, the US, and Norway included in the analyses which used different levels of aggregations, and various drivers (including several ecological and environmental covariates) to make comparisons across ecosystems.

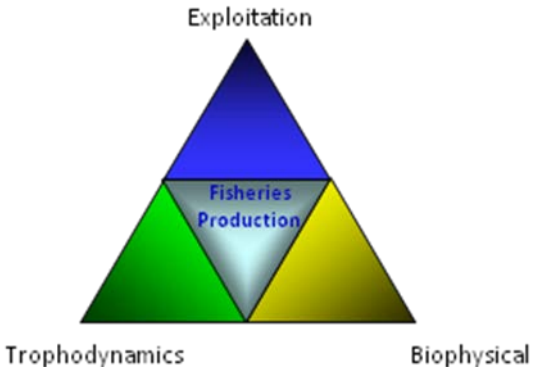


**Figure 3.15.1. The triad of drivers that influence fisheries production in ecosystems.**

Distinct, but complementary, information was provided in analyses that applied surplus production models at single-species (SS), multispecies (MS), aggregated species, and full-system levels across the 13 ecosystems.

The comparative modelling and empirical work resulted in the identification of key emergent trends and common patterns governing fishery productivity in Northern hemisphere temperate marine ecosystems. Holsman *et al*. (2012) applied models at the single species level of organization, comparing both cod and herring production across ecosystems. Lucey *et al*. (2012) applied models at intermediate levels of organization, including habitat-based groups (demersal and pelagic), size-based groups (small, medium, and large average adult size), and trophic functional groups (planktivores, zoopivores—shrimp-feeders, piscivores, and benthivores). Bundy *et al*. (2012) applied models at the full system level. In nearly all cases, a production modelling approach provided better fits to the data than the null model across all levels of aggregation. A striking result of all three studies was the general similarity of estimated BRPs across ecosystems at each level of organization. In the two studies where they were included, environmental covariates specific to each ecosystem generally improved fits to the production models at both the single species and full ecosystem levels (Bundy *et al*., 2012, Holsman *et al*., 2012). It was confirmed that full-system yield in any given ecosystem generally is less than the sum of SS yields – a result seen in a suite of other empirical and modelling studies.

### 3.15.2 Using Surplus Production Models as Operating Models in Management Strategy Evaluation

Two MSE examples have been developed in the North Pacific and Northeast United States Large Marine Ecosystems using a multispecies surplus production simulation model (MS-PROD) which incorporates predation and competitive interactions between species. The first usage of MS-PROD in an MSE context was to explore the effects of managing species by aggregate groups, specifically if certain aggregations are likely to result in stock collapses. The approach was to generate two "cartoon"

ecosystems with MS-PROD based on the Gulf of Alaska (GoA) and Georges Bank (GB) ecosystems, each comprised of 10 representative species (Gaichas *et al.*, 2012, Gamble and Link, 2012). These generated ecosystems were then considered to be 'the truth'. GoA was parameterized to emphasize predation while GB was parameterized to emphasize competition. Different levels of aggregation were then applied to each ecosystem: full system, taxonomic (groundfish, flatfish, small pelagic, elasmobranchs), habitat (demersal/pelagic), feeding guilds (benthivores, piscivores, planktivores), and size (small, medium, large), and no aggregation. Several biological reference points (BRPs; e.g. MSY and $F_{MSY}$) were estimated as production model parameters based on each level of aggregation, and then simulations were run using the estimated $F_{MSY}$ BRPs to compare the equilibrium biomass and yield among the alternate fishing strategies. One of the main results from this study was that if stocks with very different productivities were aggregated, the less productive species were at risk of collapse under aggregate fishing strategies. However, if similarly productive species were aggregated, there was little loss of yield compared to a single species strategy and the equilibrium biomasses were also comparable. Additionally, it was shown that in most cases where stocks collapsed (less than 25% of unfished biomass) reducing F so that no species collapsed usually resulted in yield levels 80% or greater of aggregate MSY except in cases where a species was much less productive than the others in the aggregate (e.g. Pacific ocean perch in GoA and mackerel in GB). This suggests that multi-objective, multispecies reference points can be created by combining a minimum biomass threshold with aggregate species yield (Figure 3.15.2), and that the yield trade-offs associated with managing for multiple objectives may not be as severe as previously thought.

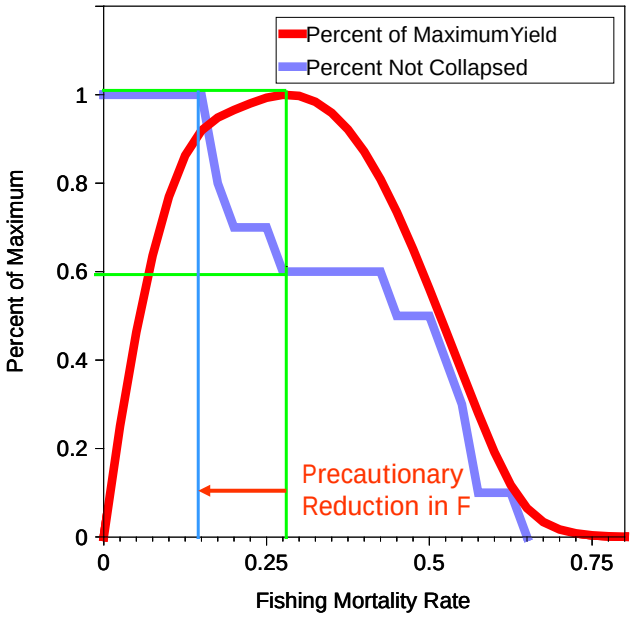


**Figure 3.15.2. An example of how to create multi-objective reference points using a yield curve overlaid with the proportion of stocks not collapsed. In this case, 60% of stocks collapse at MSY, but by reducing F, it's possible to achieve ~90% of maximum yield while preventing any stock from collapsing.**

The second usage of MS-PROD as an operating model in an MSE was the incorporation of simple climate effects (Gamble and Link, 2012). A simple modification of the NEUS MS-PROD model (Gamble and Link, 2009) explored the effect of simulating climate effects by decreasing the intrinsic rate of growth for the stocks that make up

the groundfish functional group by 10% based on other studies in the region which show that many groundfish stocks have been affected by climate change (Nye *et al*., 2009). While very preliminary, the results indicate that even this simple modification can lead to complex and non-intuitive effects such as some groundfish stocks increasing in biomass compared to the non-climate scenario although they had their intrinsic rate of growth reduced by 10%. These results were primarily due to the trophodynamic interactions between species; specifically competitive release of other groundfish that were negatively affected by the climate scenario. Additionally, the removals of biomass (from predation, competition, harvest, and climate) can all be easily calculated to provide information on the relative importance of each driver on the system, aggregate groups, and individual stocks.

### 3.15.3 ATLANTIS NEUS Removal Scenarios

Large marine ecosystems are complicated, with a large amount of connectivity (via trophic interactions) between ecological components and their environment. Ecosystem-based fishery management (EBFM) is moving from asking *why* one would do EBFM towards asking *how* to do EBFM, and recognition of the importance of these connections has increased as a result. Studies have shown that commercially important fish stocks may migrate out of regional areas and management units, but little research on the likely effects on the biotic community following such a shift has been done. Additionally it has been hypothesized that the lack of recovery of some commercially important species, despite significant management actions designed to allow them to do so, could be related to predation by or competition with other, less commercially desirable species. Again, little has been done to explore what the effects of removals of these less desirable species might have on the species of interest, a community, or even the entire ecosystem.

Scenarios where individual species or groups are removed from a system, thus simulating either migration out of an area or targeted removals to achieve a management goal, were carried out in ATLANTIS NEUS (Link *et al*., 2011) in order to elucidate some of the probable community or ecosystem responses following such removals. The species removed from the model were spiny dogfish, Atlantic cod, Atlantic herring, and seabirds. The final biomass of every functional group in the model was then compared to the base scenario to determine the effect of the removal. The results indicated that removals of species or groups from an ecosystem are unlikely to have simple effects due to the complexity of the interactions between species and their environment. Removals of spiny dogfish – often proposed to help the recovery of groundfish in the Northeast US large marine ecosystem– primarily resulted in predatory release on anadromous small pelagic fish and shrimp, with minor effects (0-10% change in biomass) on the other fish species, especially groundfish, in the system. Removing Atlantic cod resulted in minor to moderate effects on the other groundfish in the system, primarily through release of competitive effects, and the behaviour of the fisheries in relation to the change in ecosystem structure. Other functional groups were less affected in this removal scenario. Removing Atlantic herring affected both upper and lower trophic levels, primarily causing: other pelagics to increase in biomass, shrimp to decrease in biomass, complicated effects in the phytoplankton and zooplankton groups, and a decrease in baleen whales. Finally, the removal of seabirds led to a less than 1% change in any group in the model. To generalize, the propensity for indirect effects and unanticipated consequences is high in these scenarios; and therefore the use of such ecosystem models to explore the full range of options is

recommended to bound the scope of possible responses for the implementation of EBFM.

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# 4 ToR b): Report on the development of key-runs (standardized model runs updated with recent data, and agreed upon by WGSAM participants) of multispecies and eco-system models for different ICES regions (including the Baltic Sea, and others as appropriate)

## 4.1 Baltic Sea SMS

### 4.1.1 Overview

The key run for the Baltic Sea is produced with the SMS model. SMS (Lewy and Vinther, 2004) is a stock assessment model including biological interaction estimated from a parameterized size dependent food selection function. The model is formulated and fitted to observations of total catches, survey cpue and stomach contents for the Baltic Sea. Parameters are estimated by maximum likelihood and the variance/covariance matrix is obtained from the Hessian matrix.

In the present SMS analysis the following predator and prey stocks were available: predators and prey (cod), prey only (herring, sprat). The population dynamics of all species were estimated within the model.

### 4.1.2 Input data update

The stock units utilized in the present SMS analysis for the Central Baltic are: i) cod in Subdivisions 25–29+32, ii) sprat in Subdivisions 25–32, and iii) herring in Subdivisions 25–29, 32 (Gulf of Riga excluded).

The basic setup has been taken from WKMULTBAL (ICES, 2012a) and updated to include 2011 data (note, here the extended setup is given including configuration for forecasts):

- Predation mortality is estimated on the basis of constraint uniform size selection.
- Spatial distribution and overlap between the stocks is assumed constant between the years.

#### Cod and sprat

As the sprat population in Subdivisions 30 and 31 is rather low (landings are less than 5000 t in most recent years), the stock estimate is basically referring to Subdivision 25–29+32.

To estimate the predation mortality on these stocks, the cod assessment unit was adjusted accordingly, thus not considering part of the stock in Subdivision 30 and 31. Landings reported in these Subdivisions are in general less than 1% and in maximum 3.5% of the total catch from the Central Baltic. Consequently the effect of ignoring the two Subdivisions should not hamper a direct comparison between single species and multispecies assessment output. For sprat, the multi- and single species assessment units are not directly comparable, as the sprat stock in entire Baltic including subdivision 22–24 is treated as a single-stock unit in single species assessment.

#### Herring in Subdivisions 25-29 and 32

ICES stock assessment of the Main Basin herring has been made on 3 different units:

- Herring in the SD 25–29 and 32 including Gulf of Riga;
- Herring in the SD 25–29 and 32 excluding Gulf of Riga;
- Herring in the Gulf of Riga.

This has been done due to the complexity of stock structure and because the stock development trends in the Gulf of Riga and the Main basin are opposite. ACFM advice is based on assessments of herring in SD 25–29 and 32 excluding Gulf of Riga. In SMS, this configuration is used, too.

#### 4.1.3 Input data

Input data to SMS are given by quarter of the year. The data include:

1 ) Catch number
2 ) Catch mean weight
3 ) Proportion mature
4 ) Mean weight in the sea
5 ) Food consumption (ration)
6 ) M1 (residual natural mortality)

Years: 1974–2011
Stocks: Cod in Subdivisions 25–29+32
Sprat in Subdivisions 25–32,
Herring in Subdivisions 25–29+32 (excluding the Gulf of Riga),
A total of 55000 cod stomachs sampled in the period 1977–1994

For the period before 2006, data have been compiled by SGMAB (ICES, 2005). Afterwards, the data are taken from WGBFAS (ICES, 2012b for the recent update).

#### 4.1.4 Survey cpue data

Survey indices at age data were copied from ICES single species assessment (ICES, 2012b).

#### 4.1.5 Stomach contents data

Stomach content data from 1977-1992 were recompiled during WGSAM 2011 for use in SMS. Details on the recompilation are given in ICES 2011 (ICES, 2011).

A further update conducted during WGSAM 2012 concerned the use of the statistical distribution used to estimate the likelihood of the stomach content distribution. If the 2009 key run, the lognormal distribution was used whereas the Dirichlet distribution was chosen this year due to the fact that this distribution is theoretically more appropriate to modelling fractions (as diet composition).

#### 4.1.6 Modelling size preferences

In order to avoid a large increase in cod recruitment (which is not supported by independent information), the uniform preference was implemented in subsequent runs (ICES, 2012a).

### 4.1.7 Key run summary sheet

| Area | Baltic Sea |
|---|---|
| Model name | SMS |
| Type of model | Age-length structured statistical estimation model |
| Run year | 2012 |
| Predatory species | Cod |
| Prey species | Cod, herring, sprat |
| Time range | 1974-2011 |
| Time-step | Quarterly |
| Area structure | Baltic Sea |
| Stomach data | 1977-1992 |
| Purpose of key run | Making historic data on natural mortality available |
| Model changes since last key run | Used uniform size preference instead of lognormal, used Dirichlet distribution for diet composition instead of lognormal, used average temperature in herring and sprat Ricker-type recruitment models |
| Output available at | http://www.ices.dk/reports/SSGSUE/2012/WGSAM/SMS_summary.csv |
| Further details in | Report of the Working Group on Multispecies Assessment Methods 2012 |

### 4.1.8 Results

The key-run converged and the uncertainties of parameters and key output variables were obtained from the inverse Hessian matrix. A summary of results is provided in Figures 4.1.1–4.1.3.

The input and output from the model are available online as ASCII file downloads from http://www.ices.dk/reports/SSGSUE/2012/WGSAM/SMS_summary.csv

Cod biomass has continued to increase and is now on a level that is comparable to the late 1980s-early 1990s. As a consequence, predation pressure on herring and sprat has increased. Cod cannibalism has also markedly increased, since the strong year classes from 2008 and 2009 have grown to a size that allows consumption of conspecifics. Cod recruitment, however, has stagnated since 2009 and remains on a low level, so it is doubtful whether the increase of cod SSB is continuing.

Despite the increase in predatory cod biomass, the biomass of herring appears stable at a moderate level, which probably is due to relatively low fishing pressure on this stock and the relatively stable recruitment levels. Sprat biomass is decreasing, probably due to predation by cod, even if fishing mortality has decreased.

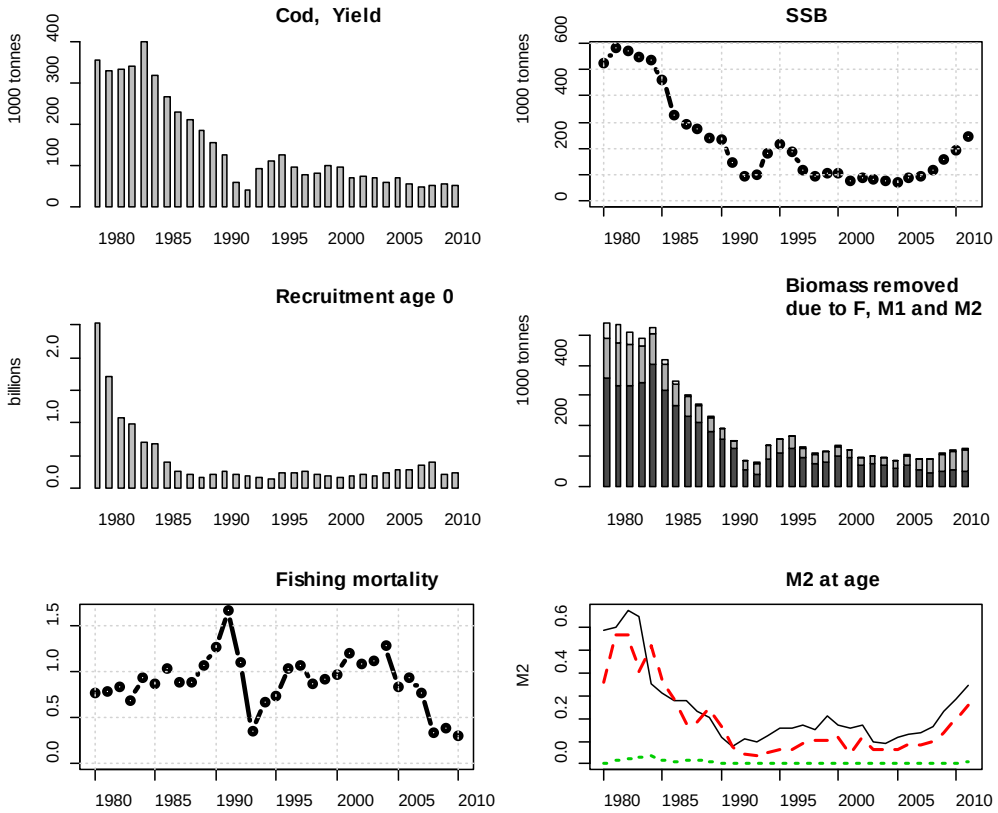


**Figure 4.1.1. SMS output for Baltic cod. Yield, SSB, recruitment, biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1), and fishing mortality. The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.**

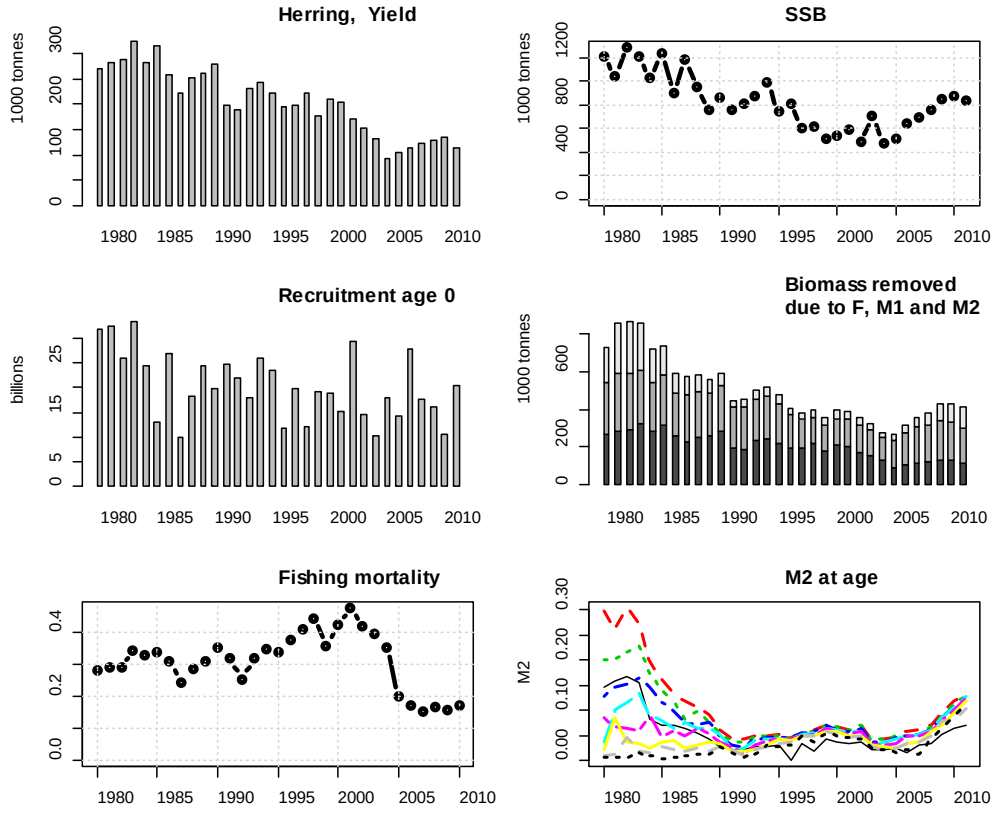


**Figure 4.1.2. SMS output for Baltic herring. Yield, SSB, recruitment, biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1), and fishing mortality. The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.**

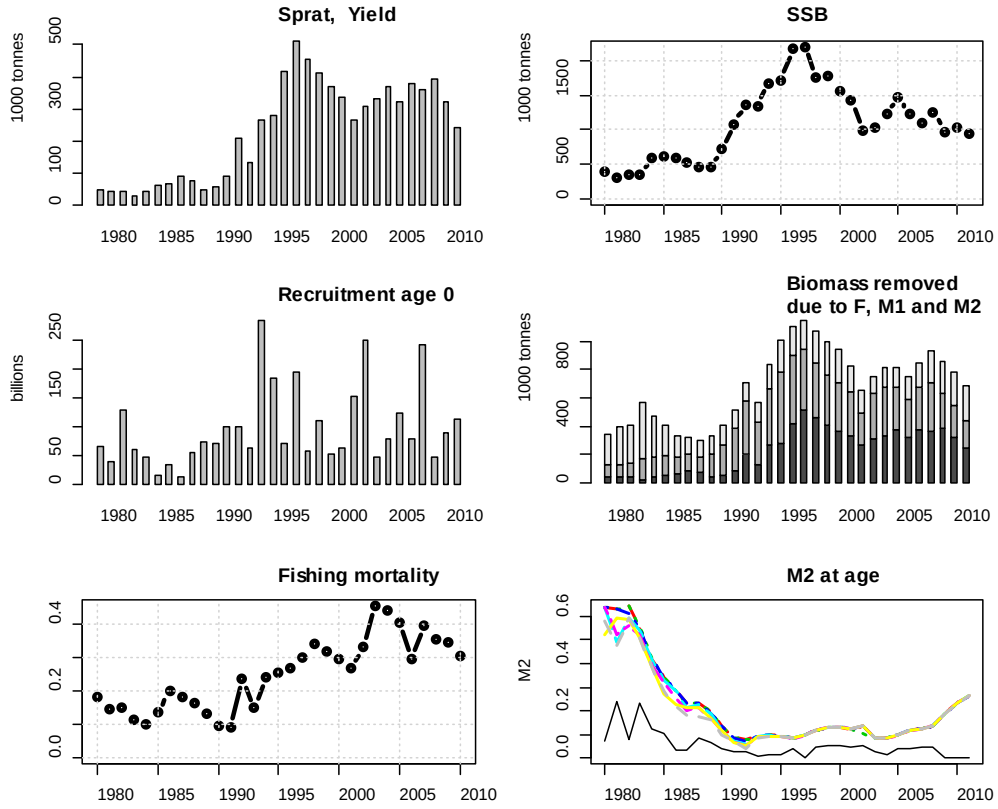


**Figure 4.1.3. SMS output for Baltic sprat. Yield, SSB, recruitment, biomass removed due to fishery (F), predation by SMS species (M2) and residual natural mortality (M1), and fishing mortality. The predation mortality (M2) presented by the 0-group (black solid line) is for the second half of the year. The M2 for the rest of the ages are annual values.**

### 4.1.9 Comparison with the 2011 (WKMULTBAL) key run

The present 2012 key run was compared to the keyrun conducted during WKMULTBAL (ICES, 2012a), as this was the most actual widely accepted update before the WGSAM meeting. The comparison is presented in Figures 4.1.4-4.1.6.

There are no major deviations. Cod biomass estimates have slightly decreased for recent years, while fishing mortality estimates have slightly increased. This is the same as observed in comparing runs from the single-species assessment working-group WGBFAS (ICES, 2012b), and is probably due to an update of the tuning data.

The spike in herring recruitment in 2008 in the 2011 WKMULTBAL run has been removed in the current keyrun. It was due to punching error in the input data.

Sprat fishing mortality is estimated slightly lower by the 2012 keyrun, as is recruitment. In consequence, there is almost no change in SSB estimates between the two runs.

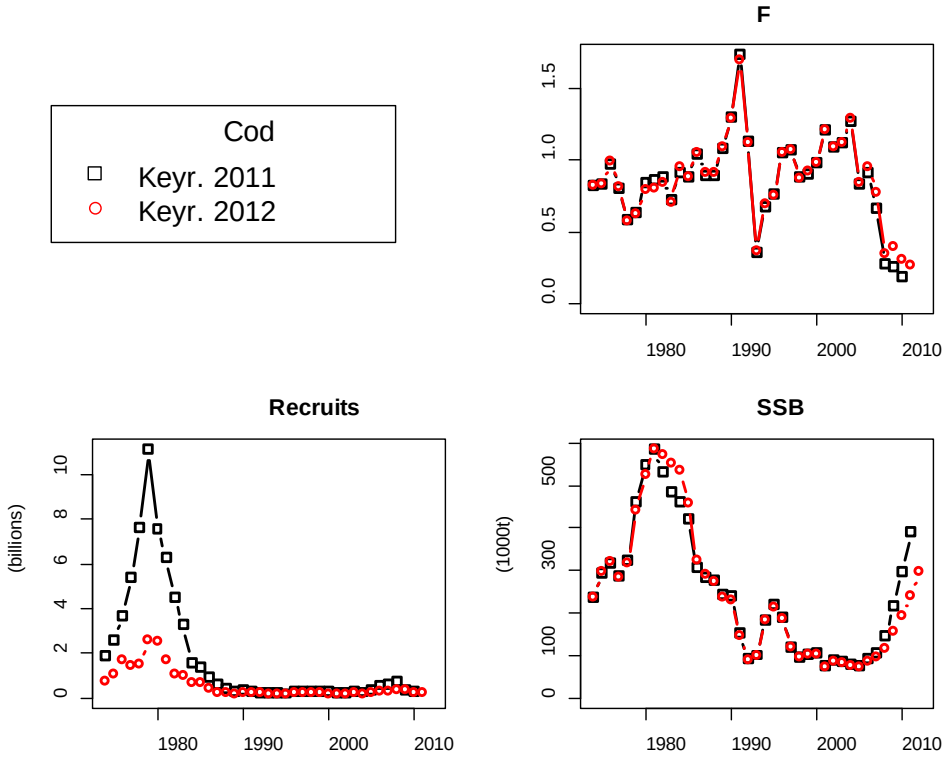


**Figure 4.1.4. Baltic cod – comparison of fishing mortality (F), number of recruits (billions) and spawning-stock biomass (SSB) estimates between the 2011 WKMULTBAL and the 2012 WGSAM keyruns.**

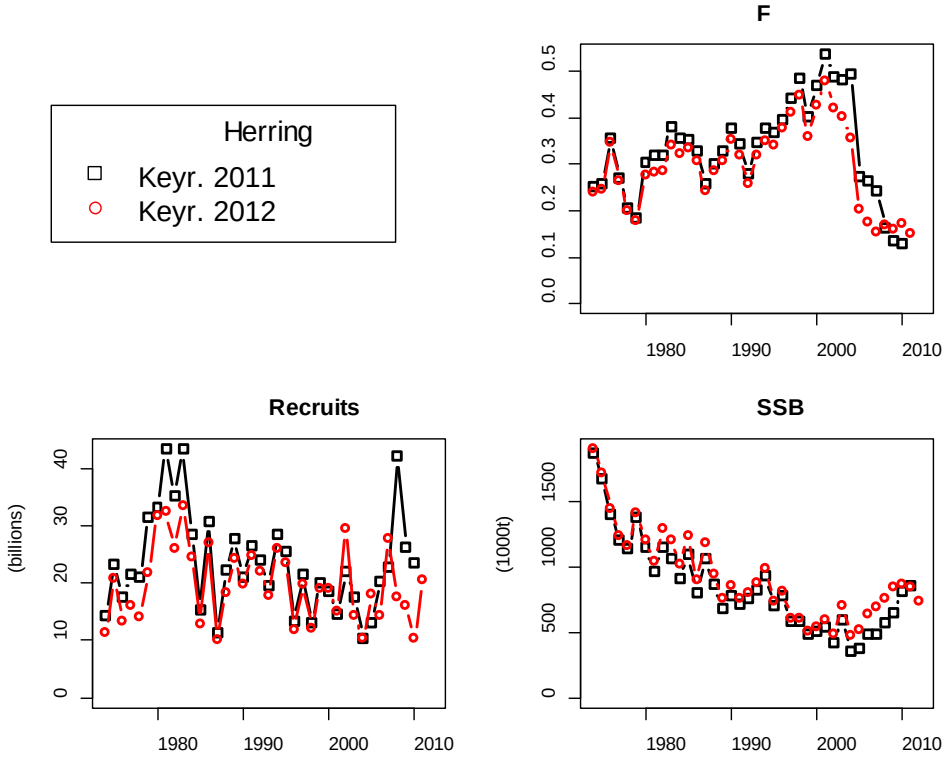


**Figure 4.1.5. Baltic herring – comparison of fishing mortality (F), number of recruits (billions) and spawning-stock biomass (SSB) estimates between the 2011 WKMULTBAL and the 2012 WGSAM keyruns.**

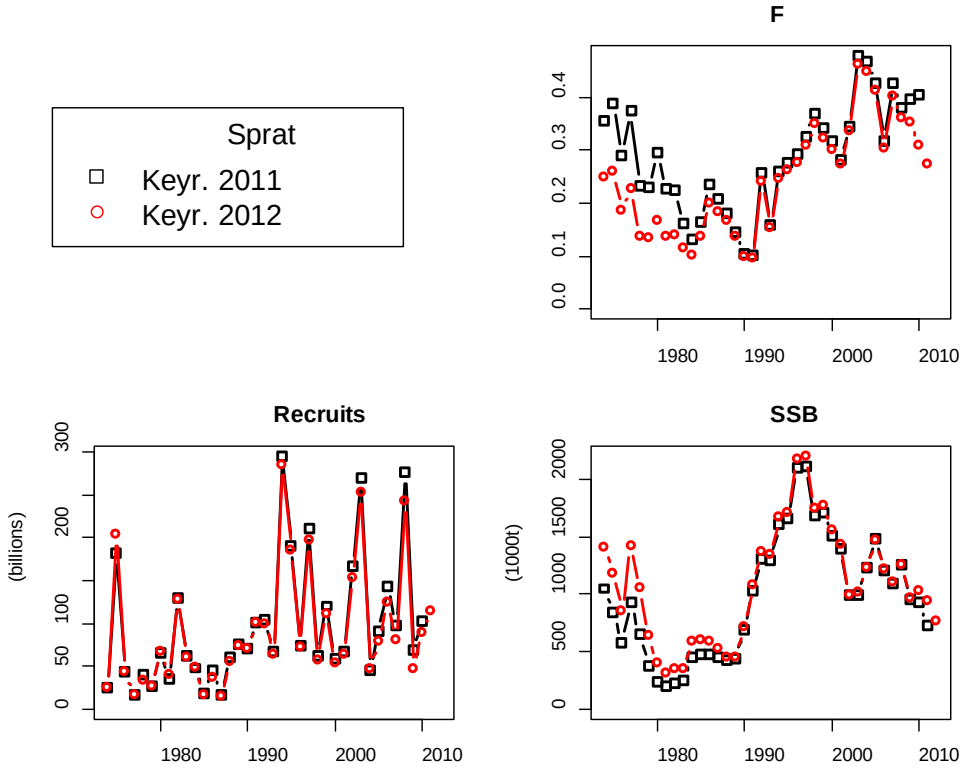


**Figure 4.1.6. Baltic sprat – comparison of fishing mortality (F), number of recruits (billions) and spawning-stock biomass (SSB) estimates between the 2011 WKMULTBAL and the 2012 WGSAM keyruns.**

### 4.1.10 Biomass eaten

The herring and sprat biomasses consumed annually by cod have increased, concurrent with the increase in cod stock size (Figure 4.1.7). The increase is more pronounced for the herring stock, because relatively more large cod are now consuming herring, whereas the smaller sprat has always been subject to cod predation, even during times of low cod biomass and size. As Figure 4.1.8 shows with the absolute biomasses consumed, cod cannibalism is hardly visible. However, even this small increase in absolute consumption causes a visible increase in young cod (ages 0 and 1) predation mortality rates.

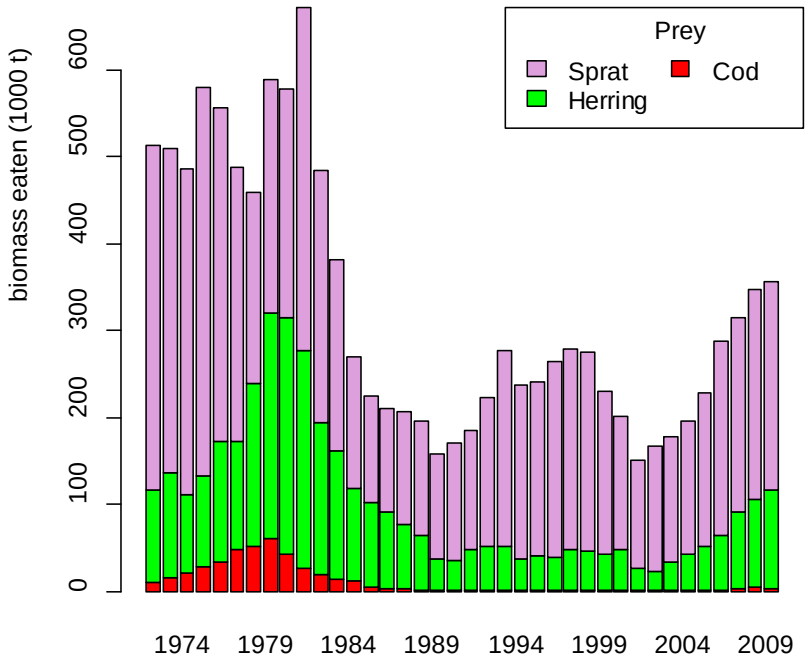


**Figure 4.1.7. Biomass of commercial fish eaten by Baltic cod (in 1000 tonnes).**

## 4.2 Tor b) (i) If needed, run additional model runs to those made by WGSAM in 2012 for the North Sea in order for fulfilling ToR b.

The group considered that updating the North Sea key run from WGSAM 2011 with one extra year of data would not significantly affect the predictions and multispecies advice as the time-series of the 2011 key run goes almost 50 years back in time.

## 4.3 Tor b) (ii) Develop draft advice text for the ICES Advice Report 2013 for a multispecies management plan for the North Sea along the lines done in 2012 for the Baltic.

This ToR is considered together with ToR i (Section 11).

## 4.4 References

ICES. 2005. Report of the Study Group on Multispecies Assessment in the Baltic. ICES CM 2006/BCC:07, 116pp.

ICES. 2011. Report of the Working Group on Multispecies Assessment Methods. ICES CM 2011/SSGSUE:10.

ICES. 2012a. Report of the Workshop on Integrated/Multispecies Advice for Baltic Fisheries. ICES CM 2012/ACOM:43.

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Lewy, P., and Vinther, M. 2004. A stochastic age-length-structured multispecies model applied to North Sea stocks. 20. 33 pp.

# 5 ToR c): Work towards implementing new stomach sampling programmes in the ICES area in the near future

Stomach sampling on a regular basis is needed not only to ensure that multi species and ecosystem models used for assessment purposes remain relevant but also to advise on the MSFD descriptor 4 regarding the structure and functioning of foodwebs (see ToR e).

There is now a study financed by the Commission to sample stomachs in the North Sea in 2013 (MARE/2012/02). However, this will only give a snapshot in time and for the North Sea only three predators (1600 mackerel stomachs, 1600 grey gurnard stomachs, 800 hake stomachs) will be sampled due to limited resources. In addition, there has been a limited stomach sampling of saithe carried out by Norway since 2009 (about 700 stomachs annually). Especially for the monitoring of foodwebs (e.g. changes in diet of predators over time) a longer term perspective is needed.

Apart from this new EU program to be started, there has only been a limited coordinated stomach sampling effort in the Skagerrak. Currently the Skagerrak (and Kattegat) seems to be, according to IBTS data, the most important nursery area for North Sea cod. Predation processes in this area are a black box, thus limiting our process understanding. Further, there is a lack of relevant information necessary to formulate simple foodweb models to provide foodweb indicators specific to this region within the MSFD and for fisheries management advice.

In the North Sea and Skagerrak, there are four main issues that make establishing a long-term sampling system difficult:

1 ) Collection of stomachs is expensive if dedicated surveys are required.
2 ) Stomach analysis is expensive and currently not financed on a regular basis.
3 ) National sampling and analyses of stomachs are not coordinated.
4 ) The North Sea (including Skagerrak) foodweb is complex and a large number of predatory species should ideally be sampled.

## 5.1 Collection of stomachs is expensive if dedicated surveys are required

This problem can be, to a large degree, addressed by sampling stomachs on existing surveys. There are discussions going on to extend the IBTS survey and other surveys to ecosystem surveys (ICES, 2012). One of the suggestions for this by WGISUR is to collect stomachs on a regular basis during these surveys.

## 5.2 Stomach analysis is expensive and currently not financed on a regular basis

The DCF financing system will change in the next years and funding of sampling and analysing stomachs may be possible via the European Fisheries Fund in future if they are a necessary prerequisite to improves fisheries management.

## 5.3 National sampling and analyses of stomachs are not coordinated

National institutes sample and analyse stomachs on their cruises to some extent. Often this occurs with a focus on project related questions, rather than with a larger

regional perspective in mind and sampling preferences differs between institutes. This leads to a situation where there is information for a predator species only from some parts of the North Sea at any given time. Drawing conclusions for the whole predator population in the North Sea from these data are very difficult. If these efforts could in some way be coordinated to ensure wider spatial and temporal coverage, these data could be valuable input to the models. A better coordination between national institutes would enhance the information derived from stomach sampling without adding a need for extra resources.

The most obvious place for the coordination is most likely in the IBTSWG and WGBEAM. However, national institutes could also bilaterally contact other national institutes. The institutes sampling various parts of the North Sea can be seen in Figure 5.3.1.

Given substantial financial cuts in various institutes, even simply better coordination of stomach sampling can be a preliminary solution. Any real long-term stomach sampling plan needs funding, especially for analysing stomachs in the lab.

## 5.4 The North Sea (including Skagerrak) food web is complex and a large number of predatory species should ideally be sampled

Not all predatory fish are routinely sampled across age-classes, and hence sampling presents a significant amount of additional work on-board the surveys (but see Link *et al.*, 2008). This problem could be ameliorated by sampling only to a selection of species in each year, with a rolling program to sample a specific species every 3–5 years or so. It would make sense to establish a rolling scheme, similar to the scheme for biological sampling of additional species during the NS-IBTS Q, where in a certain year all national institutes that are interested in collecting stomachs focus on 1–2 predator species. In the next year(s) then other species will be collected. Such a system would be beneficial for everyone as the different interests of the institutes are served and the data basis as well as the coverage in space and time is improved.

To cover the whole North Sea during the IBTS in each of the two quarters (Q1 and Q3) at minimum one of each national institutes covering a square in the maps (Figure 5.3.1) is needed. In the third quarter the Dutch could contribute to the samples by their beam trawl survey covering western North Sea (ICES, 2012, WGBEAM) potential other institutes might be capable of contribution by national surveys.

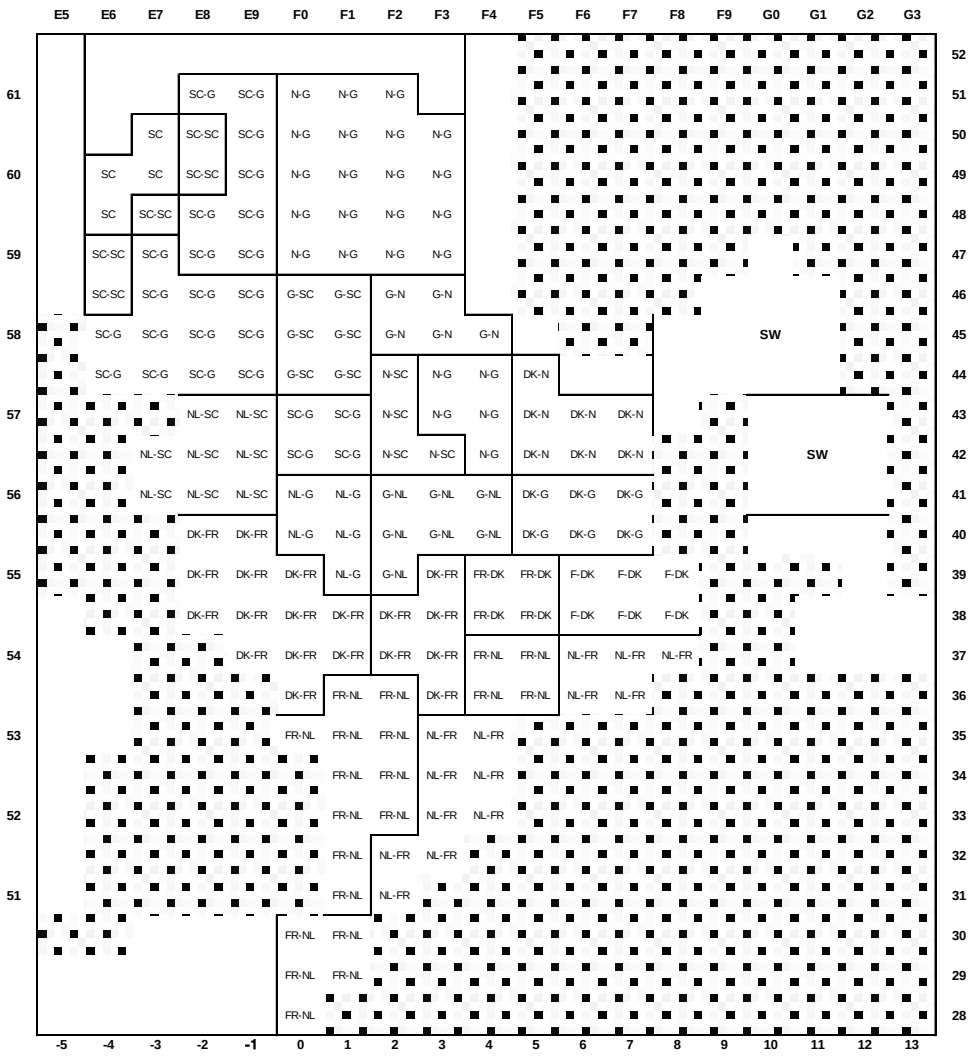


**Figure 5.3.1. Map indicating the spatial coverage of different countries in the quarter 1 IBTS.**

## 5.5 Methods

A detailed manual with best practices has been published as 'Manual for ICES Stomach sampling projects in the North Sea and Baltic Sea' (ICES, 2010). Stomach sampling should follow this manual where possible.

For the North Sea this means that haddock, whiting, saithe, gurnard, horse mackerel, mackerel, starry ray, turbot and brill, with John Dory and hake as optional extensions, should be collected. The manual recommends sampling 5 rather than 10 stomachs per 5 cm size group of each predator, with the exception of saithe, mackerel and horse mackerel, where a large proportion of the stomachs are empty. For these species, 15 stomachs should be sampled from each size group.

The necessary taxonomic resolution and details of the requested data means that most of the stomach could be analysed on board already. This would reduce the largest part of the extra costs of sampling stomach on the regular surveys.

## 5.6 Requests

The regular collection of stomachs is an option to improve the information on the ecosystem from surveys coordinated by IBTSWG and WGBEAM. WGSAM would

like to request IBTSWG, WGBEAM and WGISUR to endorse the request for stomachs to be collected on the surveys and if possible to analyse a part of these stomachs on board. If not, to then specify the main issues for not doing this, and specifically identify extra finances needed to do so.

If these groups can endorse the request, a next step is the construction and maintenance of the database to store these types of data. At the moment the stomach data from the year-of-the-stomach (ICES, 1991) are available via the website of the ICES-datacenter. The plan of the EU MARE-project is to extend this database, such that it can contain additional stomachs. When stomach collection becomes routine on the surveys it would mean that the stomach-database should be running similarly to the DATRAS-database.

## 5.7 Baltic

A new stomach sampling and stomach analyses programme is currently initiated in the Baltic Sea. The predator species sampled in the Baltic Sea are cod and whiting (in those areas where those species occur). The objectives of the programme are to:

- incorporate all appropriate historical stomachs content information into the Baltic stomachs content databases
- conduct stomach content analyses of new cod stomachs collected in the Baltic Sea, to support our knowledge of the spatial and temporal stability of cod preferences
- conduct stomach content analyses of whiting stomachs collected in the Baltic Sea to support our knowledge of potentially important predators for which the diet is currently poorly known

The overall goal is to sample ten stomachs per 1 cm size group of cod and whiting from each ICES Subdivision (SDs 21 – 26 and 28) in each quarter. The ten stomachs may include empty stomachs, however, stomachs that are obviously regurgitated are discarded (see below for details on which stomachs to select).

Within each SD, a wide geographical coverage of samples should be obtained whenever possible. In those SDs with many stations and where large catches can be expected (SDs 21–26 for cod and SDs 21–24 for whiting), this can be achieved by limiting the sampling frequency to (a) no more than 2 stomachs per cm group per haul, and (b) no more than 2 stomachs per cm group per day.

However, in those SDs with fewer stations and/or where catches may be relatively small (SD 28 for cod and SD 25 for whiting), a higher sampling frequency per haul/day may need to be chosen in order to obtain sufficient numbers of samples per length group.

As most SDs are covered by several countries, it will be agreed by e-mail correspondence in advance of the surveys how many samples per length class each country is taking in which SD.

Stomachs will be selected randomly within each length group. To assure random selection within size classes, and as additional single-fish data are needed anyways, it is recommended to use fish selected for single-fish data, otolith and maturity sampling whenever possible.

The samples should be frozen individually in plastic bags including a label describing the sampled fish (Table A1). Every fish needs to have a unique ID number, which is noted on the label and which makes it possible to connect the stomach to the other

relevant single-fish, haul and environmental (CTD) data. Stomachs are collected from cod/whiting > = 15 cm. To ease the handling, cod and whiting < 15 cm are frozen as a whole (2 specimen per cm group and per haul).

The fish sampled for stomachs will be selected with care to obtain reliable data and the stomachs should be categorized as follows:

1 ) **Everted stomachs.** Some fish have everted stomachs. Since it is not known whether these stomachs contained food or not, such fish must not be used for stomach sampling.

2 ) **Regurgitated stomachs.** Some fish have regurgitated all or part of their stomach contents and these fish must not be collected for analyses. However, the number of regurgitated stomachs encountered during the examination must be recorded to ensure that the proportion of feeding fish in the sample is accurately defined. In practice, it is often difficult to tell whether regurgitation has taken place, but in situations where the stomach is flaccid or distended, but contains little food, experimental work by Robb (Robb 1992) indicates that the size of the gall bladder is a useful practical indicator of the recent feeding history of the fish. A large densely coloured gall bladder indicates that a stomach has been empty for some time and has not recently lost its content by regurgitation. The criterions are summarized in Table A2 and should be applied when assessing whether a stomach should be classified as regurgitated or empty.

3 ) **Stomachs of feeding fish showing no signs of regurgitation**. These should be collected for analyses. It should be noted that not all feeding fish have grossly distended stomachs, i.e. feeding does not necessarily mean full.

4 ) **Empty stomachs**

5 ) **Stomachs with only indigestible skeletal remains** (polychaete bristles, mollusc shells and opercula, fish bones and otoliths etc).

The material collected at sea should originate from feeding fish showing no evidence of regurgitation and from non-feeding fish.

Categories 1 and 2 are not collected, but it should be noted how many stomachs were in these conditions. The sampling should continue until one stomach classified as either empty, containing only skeletal remains or feeding is obtained. The state of the gall bladder should be recorded using the scale in Table A2.

For each stomach, the following corresponding single-fish data will be collected:

**Minimum:**

1 ) total length

2 ) weight

3 ) gutted weight

4 ) sex

**Optimum (additionally):**

5 ) maturity stage

6 ) gonad weight

7 ) liver weight

8 ) age/otolith samples

Each stomach can be connected to the respective haul information. As a minimum, the following information should be collected:

1 ) Gear type/Fishing method (see Table A3)
2 ) Date
3 ) Start time
4 ) Start Position
5 ) Depth

Station information should be recorded in ICES exchange format to be consistent with information uploaded to DATRAS.

Each stomach can also be connected to the corresponding CTD profiles.

As a minimum, the following information should be collected:

1 ) Temperature
2 ) Salinity
3 ) Oxygen content

**Table A1. Label to be included with each stomach sample.**

| ICES Stomach sampling programme |
|---|
| Ship |
| Cruise number |
| Haul number |
| Date |
| Rectangle |
| Species |
| Size |
| Gall bladder Class (Table 2) |
| Fish no |

**Table A2. Condition of gall bladder and hind guts used to differentiate between empty and re-gurgitated stomachs.**

| Stage | Gall bladder | Bile colour | Hind gut | State |
|---|---|---|---|---|
| 1 | Shrunken, empty or with small amount of bile | Pale | Contains large amounts of bile and digested food material | Feeding* |
| 2 | Elongate | Pale green to light emerald green | Contains some bile and digested food particles | Feeding* |
| 3 | Elongate | Dark green | Empty or contains some food particles | Empty |
| 4 | Round | Dark blue | Empty | Empty |

***If fish satisfying these criteria are found without food in their stomach they should be classified as regurgitated.**

**Table A3. Fishing method codes.**

| Method code | Description |
|---|---|
| DEM | Demersally caught by trawling or seining gears |
| PEL | Pelagically caught by trawling or seining gears |
| DHL | Demersal hook and line |
| PHL | Pelagic hook and line |
| DGN | Demersal gillnets |
| PGN | Pelagic gillnets |

**Table A4. Species coordinators. Bold indicates main contact point.**

| Predator | Stomach analysis at |
|---|---|
| Baltic Sea cod | DTU-Aqua, NMFRI Poland, BIOR Latvia |
| Baltic Sea whiting | DTU-Aqua |

As some SD's are covered by several nations, a slight over-sampling in those SD's (i.e. SDs 25 and 26) will be implemented to ensure sufficient spatial coverage and enough stomachs for all size classes. The maps in Figure 5.7.1 show the trawl stations of all participating nations, which indicate which nation should be taking how many cod and whiting stomachs in which area. The different nations will sample the following numbers of stomachs per 1 cm length class:

| COD | SD 22 | SD 24 | SD 25 | SD 26 | SD 28 |
|---|---|---|---|---|---|
| **Denmark** | 8 | | 10 | 2 | |
| **Germany** | 2 | 10 | | | |
| **Sweden** | | | 5 | | 5 |
| **Poland** | | | 5 | 5 | |
| **Latvia** | | | | 5 | 5 |
| **Lithuania** | | | | 3 | |

| WHITING | SD 22 | SD 24 | SD 25 | SD 26 | SD 28 |
|---|---|---|---|---|---|
| **Denmark** | 5 | | 5 | | |
| **Germany** | | 5 | | | |
| **Sweden** | | | 5 | | |
| **Poland** | | | 5 | | |
| **Latvia** | | | | | |
| **Lithuania** | | | | | |

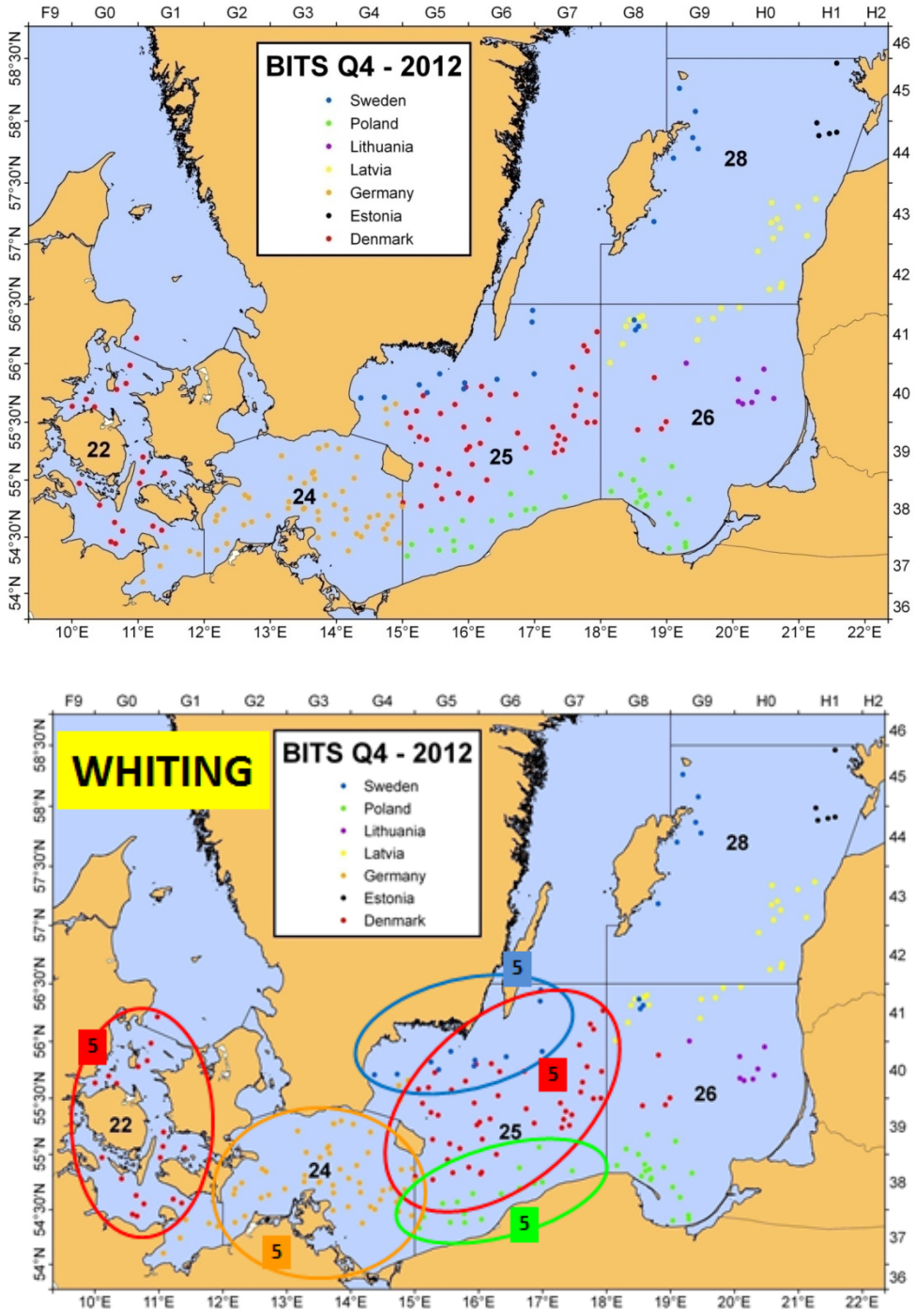


**Figure 5.7.1. Maps showing the sampling effort by nation (number of stomachs per cm length group) for cod and whiting scheduled for the BITS fourth quarter survey in 2012.**

## 5.8 References

ICES. 1991. Manual for the ICES North Sea stomach sampling project in 1991. ICES C.M. 1991/G:3.

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Link, J.S., Burnett, J., Kostovick, P., and Galbraith, J. 2008. Value-added sampling for fishery independent surveys: Don't stop after you're done counting and measuring. Fish. Res., 93:229–233.

Robb, A. P. 1992. Changes in the gall bladder of whiting (*Merlangius merlangus*) in relation to recent feeding history. ICES J. Mar. Sci. (1992) 49 (4): 431–436.

# 6 Tor d): Explore how 'virtual multispecies datasets' (including survey, catch and stomach content data) for use in multiple multispecies models, especially for comparison and sensitivity testing, could be constructed

## 6.1 Utility of virtual vs. real datasets

When selecting datasets to use for developing and validating models, either real or "virtual" (simulated) datasets can be used. Each approach has strengths and weaknesses, and it is important to recognize the limitations of the approach being used.

### 6.1.1 Real datasets

Using real-world data gives an assurance that the data are representative of actual dynamics (including autocorrelation, observation and process error), but the true error structure and parameters (e.g. stock biomass) are typically unknown. Although the datasets cannot be tweaked to give different scenarios, a suite of different cases can be obtained by using examples from a range of different ecosystems. As the "truth" is not known, it can be difficult to judge relative performance between different models if comparing historic trajectories.

### 6.1.2 Virtual datasets

The advantages of using virtual datasets are that the "truth" is known and that a range of formulations of process and observation error can be added to this truth. The disadvantage is that the simulation model or the selected method to add process and observation error may not fully capture the dynamics of the real system to be modelled. Model simplifications and misspecifications may limit the ability of the simulated datasets to replace real world data. Furthermore if a model fails to capture the dynamics in a simulated dataset it is not clear if the failure lies with the model being examined or the one that created the dataset. In addition it is dangerous to simulate data using a similar type of model to that which will be tested on the data. In such a case the "validation" exercise is reduced to a check on the consistency of the model, rather than its ability to capture real world dynamics.

Some of these pitfalls in using artificial data can be avoided by creating a range of datasets from structurally different generating models. This gives a much more robust suite of tests than simply adding different error structures to a single model run. A second potential advantage of using simulated data are that long time-series are more readily available. This makes it easier to test the impact of the length of the dataseries on assessment model performance.

### 6.1.3 Evaluation criteria

A distinction must be made relating to which performance is to be evaluated. Is the goal of the exercise to evaluate the ability of the model to accurately predict historical data (hindcast) or to accurately predict short-term population trends? In the case of a historical (hindcast) fit, only simulated data can be used. Attempting to use real world data can produce a situation where it is impossible which of the models being tested has performed "best" in a hindcast, since the data may not be a true representation of the population. Either simulated or real data can be used to make short-term predictions and the models can then be evaluated by comparing e.g. the predicted biomass or landings to observed survey indices or landings.

### 6.1.4 Multispecies datasets

The above issues arise for both single and multispecies datasets, with multispecies datasets being essentially a collection of single species datasets, plus data on diet and food availability. However, there are additional difficulties in generating stomach content/diet data in multispecies contexts, which are discussed below.

## 6.2 Summary of previous work at WGSAM

The issue of virtual multispecies datasets has been discussed in previous WGSAM meetings, and a full report is present in the WGSAM 2011 report (ICES, 2011). The key requirement is that data should be independent of the models to be tested, internally consistent and biologically realistic.

The 2011 report suggested that a first attempt at such a dataset could include two forage fish, one predator on these, one apex predator, and possibly other non-target species. For each of these, typical single species datasets would be required together with consumption information. In multispecies datasets particular issues arise with modelling stomach contents. Multispecies models aggregate over large areas (often with no area structure) and long-time-steps (typically monthly or quarterly). Real world stomach contents are highly variable at small spatial and temporal scales. Stomach contents will vary based on the recent feeding history of the fish, and overlap or not between predator and prey can be a matter of metres and hours (as opposed to the months and 100s of km in typical models). Furthermore the stomach contents are a representation of the recent feeding history compounded by differential digestion rates for different prey. Consequently multispecies models cannot, in general, produce virtual stomach datasets that approximate to those encountered in the real world. A simplification is that models can produce known predation mortalities. This can be used as "data" in a first step in testing models, although this presents the models to be tested with an unrealistically easy test.

Furthermore the range of available multispecies models is rather limited, thus making the necessary task of finding classes of models to create the simulated data that are different from those being tested difficult. WGSAM 2011 discussed possible solutions to this, from a simplified spreadsheet generating the data, through to extracting data from a full ecosystem "Atlantis" model. None of these approaches were considered to be ideal, although Atlantis was considered to have potential.

Two possibly useful models to generate virtual datasets are the PDMM (Shephard *et al.*, 2012) and the FCSRM (Hartvig *et al.*, 2011; Rossberg, Houle and Hyder, unpublished). Both models can represent relatively large communities (>20 species) of competing and trophically interacting fish, and in both models the mechanizms regulating population sizes (and so determining, e.g. MSY) emerge inherently from model dynamics rather than being hard coded. They are structured very differently from models used in multispecies assessment, and so reduce the risk of circularity in model testing. An advantage of the PDMM is that it represents entire marine communities with 1000s of species and 100s of fish populations; an advantage of the FCSRM is that it explicitly represents population size structure and is easily modified to represent age structure as well.

Yet another option is to use a simplified system such as the Baltic and compare multispecies methods based on their ability to forecast either catch or survey indices in this system. This would provide the models with equal potential to perform well and would thus represent the 'fairest possible' comparison. However, the task of setting up the models is not trivial and would require substantial work.

## 6.3 Relation to work conducted at WGMG/conference on stock assessment methods for sustainable fisheries

The previous section discussed the issue of providing multispecies datasets to test multispecies models. However a second area of interest is in using multispecies models to produce datasets for use in testing single species assessment models. This relates to the ongoing work, including ICES WGMG (the Methods WG) work in 2012 on producing a series of single species datasets for use in validating single species models. These will be used at the workshop on "Stock assessment methods for sustainable fisheries" in 2013. It is important that these generating models are different from those to be tested. It should be stressed here that using simulated data from one model to "validate" a model of similar structure and assumptions is essentially worthless.

One potential area for cooperation between WGSAM and the work on single species datasets is the use of multispecies models to generate single species virtual datasets, for use in testing single species models. This is to some extent being done, with some of the JRC's a4a Initiative (Assessment for All, https://fishreg.jrc.ec.europa.eu/projects) datasets to be included into FLR coming from the multispecies Gadget model. This approach has a number of different advantages:

1 ) It widens the number of models available for generating data, allowing for multiple relatively independent datasets to be produced. This is important as simply adding different error structures to a single model output does not give a robust set of test datasets.

2 ) Multispecies models generally employ a different class of model compared to the single species assessment models to be tested. This allows independent datasets with the full range of typical single species concerns (cpue trends, changes in catchability, etc.) to be created.

3 ) Multispecies models can create datasets with a greater range of "challenges" for single species assessment models. In these cases the single species models must cope with issues that can arise in the real world, but which may prove problematic for single species models. Examples include:

   a ) Variable predation induced mortalities (variation in M).

   b ) Misestimation of mortalities (bias in M)

   c ) Misidentification of species in surveys or mixed fisheries

   d ) Environmental impacts across multispecies systems

## 6.4 References

Hartvig, M., Andersen, K. H., Beyer, J. E. 2011. Food web framework for size-structured populations. J. Theor. Biol. 272 (1), 113–122.

ICES. 2011. Report of the Working Group on Multispecies Assessment Methods. ICES CM 2011/SSGSUE:10.

Shephard, S., Fung, T., Houle, J. E., Farnsworth, K. D., Reid, D. G., Rossberg, A. G. 2012. Size-selective fishing drives species composition in the Celtic Sea. ICES Journal of Marine Science, 69 (2): 223–234.

# 7 ToR e): Develop food web and ecosystem indicators (descriptor 4) relevant to the marine strategy framework directive (MSFD) from outputs of model key runs and input data

## 7.1 Indicators considered

Throughout Europe, considerable effort is being given to the development of ecosystem indicators for measuring the state of the marine ecosystems and the impact of human change upon them. Principal examples among these are (i) ICES/JRC guidelines for establishing ecosystem indicators used assessing Good Environmental Status under the Marine Strategy Framework Directive ((COM(2010)477), (ii) Ecosystem indicators defined under the EU Data Collection Framework (DCF http://datacollection.jrc.ec.europa.eu/dcf-marine/variables), and (iii) OSPAR Ecological quality objective indicators http://www.ospar.org/content/content.asp?menu=00690302200000_000000_000000

(iv) IndiSeas international working group (www.indiseas.org, Shin *et al*., 2010). Many overlaps exist among the indicator lists.

The DCF indicators and also several of the MSFD indicators are particularly relevant to fisheries management because they aim to provide measures of how ecosystems respond to fishing. Focusing on these indicators, WGSAM considered how multispecies and ecosystem models, as well as data from diet studies, can be used to calculate historical trends of indicators and make predictions of how indicators might respond to changes in management strategies. The added value of these indicators to single species assessment of fisheries management plans is the ability to identify and evaluate the trade-offs between ecological and fishery targets. This is particularly important when objectives for fisheries must be considered within the broader context of objectives for environmental sustainability.

In particular, WGSAM propose a series of indicators that relate directly to MSFD and DCR indicators and can be routinely calculated from Key Runs of multispecies and ecosystem models. Of special importance is the provision of indicators of foodwebs (MSFD descriptor 4). A brief summary of the development on this descriptor follows.

The JRC/ICES Task Group requested an approach to implement the Descriptor 4 (Foodwebs) of the MSFD provided a report in April of 2010 (Rogers *et al*., 2010). Descriptor 4 states *that "All elements of the marine foodwebs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full re-productive capacity."* Key attributes of the descriptor are considered to be: (1) Energy flows in foodwebs, and (2) Structure of foodwebs (size and abundance). For each of the attributes, evaluation criteria have been suggested:

(1a) Production or biomass ratios that secure the long-term viability of all components

(1b) Predator performance that reflects long-term viability of components

(1c) Trophic relationships that secure the long-term viability of components

(2a) Size: Proportion of large fish maintained within an acceptable range

(2b) Abundance /distribution: maintained within an acceptable range

Since then, the European Commission has published a decision on criteria and methodological standards on good environmental status of marine waters (Commission, 2010). The EU Decision states that in order to assess energy flows in foodwebs, three aspects are to be covered: (1) Productivity (production per unit biomass) of key species or trophic groups, (2) Proportion of selected species at the top of foodwebs, and (3) Abundance/distribution of key trophic groups/species spatially resolved to region, subregion or subdivision.

From the indicator list proposed by WGSAM (Table 7.1.1), 3 are considered ready for providing advice to ICES assessment working groups. The remaining indicators are requested to be reviewed by WGECO and subject to further consideration pending the outcomes of the commission review of member state proposals for indicators to satisfy MSFD. Those proposals were due for submission to the Commission on 15 October 2012. The work on foodweb indicators by WGSAM has relevance to several ICES WGs and the STECF WG on the Ecosystem Approach to Fisheries (JRC. 2012).

**Table 7.1.1. List of foodweb and ecosystem indicators WGSAM considers useful for advice of the ecosystem approach to fisheries under the MSFD to elucidate GES. The table, identifies the connection between suggested MSFD features and WGSAM suggested indicators for (1) Biodiversity, (3) Commercial fish and shellfish, (4) Foodwebs, along with the models currently available in WGSAM to generate such indicator outputs. Indicators suggested for analyses in 2012 are in bold.**

| MSFD code | MSFD Indicator (or attribute) | Indicator proposed by WGSAM | Brief explanation (see Section below for methods) | Which model currently run by WGSAM can provide this information. |
|---|---|---|---|---|
| Descriptor 1 - Biological Diversity | | | | |
| 1.1.1 | Distributional range | | | |
| 1.1.2 | Distribution within the latter, where appropriate | | | |
| 1.1.3 | Area covered by the species (for sessile/benthic species) | | | |
| 1.2.1 | Population abundance and or biomass, as appropriate | | | EwE for all, Gadget, SMS, and Stocobar for assessed species |
| 1.3.1 | Population demographic characteristics (e.g. body size or age-class structure, sex ratio, fecundity rates, survival/mortality rates) | Total fisheries-relatedand natural mortality of fish species (F+M), in practice only assessed stocks. | Management must respond to changes in F+M, not only F. | Gadget, SMS, EwE, Stocobar |
| 1.3.2 | Population genetic structure, where appropriate | | | |
| 1.4.1 | Distributional range | | | |
| 1.4.2 | Distributional pattern | | | |
| 1.5.1 | Habitat area | | | |
| 1.5.2 | Habitat volume, where relevant | | | |
| 1.6.1 | Condition of the typical species and communities | | | |
| 1.6.2 | Habitat - Relative abundance and/or biomass, as appropriate | | | |
| 1.6.3 | Physical, hydrological and chemical conditions | | | |

| | | | | |
|---|---|---|---|---|
| 1.7.1 | Composition and relative proportions of ecosystem components (habitats and species) | | | EwE, Gadget, SMS and Stocobar for assessed species |
| 1.7.1 | Composition and relative proportions of ecosystem components (habitats and species) | Gini-Simpson diversity index (species dominance) of large fish and of small fish by biomass. | Measures community change. Responds to fishing, because sensitive of abundance of the few most abundant species. | SMS for assessed species, EwE, Gadget but misses important biomass of e.g. polar cod |
| 1.7.1 | Composition and relative proportions of ecosystem components (habitats and species) | Gini-Simpson dietary diversity of each fish species. | Measures community change. Responds to fishing, because sensitive of abundance of the few most abundant species. | SMS for assessed species, EwE, potentially also Gadget. |
| 1.7.1 | Composition and relative proportions of ecosystem components (habitats and species) | Pelagic/demersal and fish/benthos | Measure changes in community structure as an indicator of the distribution of energy in the ecosystem. | EwE, Gadget, SMS and Stocobar pelagic/demersal for assessed species. |

| | | | | |
|---|---|---|---|---|
| 1.7.1 | Composition and relative proportions of ecosystem components (habitats and species) | Average Trophic Level (TL) of community | Measure of state of how the energy in the foodweb is distributed. | EwE without further information, Gadget, SMS and Stocobar for assessed species given estimates of TL. |
| 1.7.1 | Composition and relative proportions of ecosystem components (habitats and species) | Mean TL of the catch | Measure of how average trophic level of the fish caught (also known as MTI and subject to much debate) | EwE without further information, Gadget, SMS and Stocobar for assessed species given estimates of TL. |
| Descriptor 3 - Populations of commercially exploited fish and shellfish | | | | |
| 3.1.1 | Primary indicator: Fishing mortality (F) | | | Gadget, SMS and Stocobar for assessed species |
| 3.1 | Level of pressure of the fishing activity | Community F (catch/biomass) | Measure of the overall pressure on the exploited part of the ecosystem | EwE for all, Gadget, SMS and Stocobar for assessed species. |
| 3.1.2 | Secondary indicator: Ratio between catch and biomass index (hereinafter catch/biomass ratio) | | | |
| 3.2.1 | Primary indicator: Spawning-stock biomass (SSB) | | | Gadget, SMS and Stocobar for assessed species |
| 3.2.2 | Secondary indicator: Biomass indices | | | |
| 3.3.1 | Proportion of fish larger than the mean size of first sexual maturation | | | Gadget, SMS and Stocobar for assessed species |

| | | | |
|---|---|---|---|
| 3.3.1 | Proportion of fish larger than the mean size of first sexual maturation | LFI (proportion of biomass of individuals larger than a certain L threshold) | Gadget for modelled species, SMS similar (15-40 cm=small). Stocobar for cod. Size spectra model potential. All require decisions on lower cut-off and L |
| 3.3.1 | Proportion of fish larger than the mean size of first sexual maturation | Slope, amplitude of trophic cascades, non-"linearity" of size spectrum. | Gadget, SMS, Stocobar with mean length-at-age for cod: yes for slope and non-linearity of size spectra |
| 3.3.1 | Proportion of fish larger than the mean size of first sexual maturation | 95%-tiles of length of individuals in fish community. | Gadget for modelled species, SMS similar, Stocobar for cod. Size spectra model potential. All require decisions on lower cut-off. |
| 3.3.1 | Proportion of fish larger than the mean size of first sexual maturation | LSI (proportion of biomass belonging to species larger than a certain Linf threshold) | Gadget for modelled species, SMS similar. Size spectra model potential. All require decisions on lower cut-off. |
| 3.3.2 | Mean maximum length across all species found in research vessel surveys | | |
| 3.3.3 | 95% percentile of the fish length distribution observed in research vessel surveys | | |

| | | | | |
|---|---|---|---|---|
| 3.3.4 | Size at first sexual maturation, which may reflect the extent of undesirable genetic effects of exploitation (secondary indicator) | | | |
| Descriptor 4 - Foodwebs | | | | |
| 4.1.1 | Performance of key predator species using their production per unit biomass (productivity) | | | Gadget, SMS and Stocobar for assessed species |
| 4.1.1 | Performance of key predator species using their production per unit biomass (productivity) | Total (F+M) and natural mortality of fish species, in practice only assessed stocks. | | Gadget, SMS, EwE, Stocobar |
| 4.1.1 | Performance of key predator species using their production per unit biomass (productivity) | Mean weight of predatory species | Measure of condition of predators relating to food availability | SMS (Baltic, North Sea potential), Gadget (potential), Stocobar (cod only) |
| 4.1.1 | Performance of key predator species using their production per unit biomass (productivity) | Loss in secondary production resulting from fishing. (L index) | Responds predictably to disturbance of community by fishing mortality, but also to changes in primary productivity. | EwE |
| 4.1.1 | Performance of key predator species using their production per unit biomass (productivity) | Gini-Simpson dietary diversity of each fish species. | | SMS for assessed species, EwE, Gadget potentially. |

| | | | | |
|---|---|---|---|---|
| 4.1.1 | Performance of key predator species using their production per unit biomass (productivity) | Mean transfer efficiency for a given TL or size. | Important for transport of energy to higher trophic levels. | EwE, size spectra potentially |
| 4.1.1 | Performance of key predator species using their production per unit biomass (productivity) | Average recruitment anomaly | | Only hindcast currently: Gadget, SMS, Stocobar |
| 4.2 | Proportion of selected species at the top of foodwebs | Slope, amplitude of trophic cascades, non-"linearity" of size spectrum. | Responds predictable to disturbance of community by fishing mortality.<br>Non-"linearity" characterizes trophic efficiency.<br>Trophic cascades, because minima can lead to species loss. | Gadget, SMS, Stocobar with mean length-at-age for cod: yes for slope and non-linearity of size spectra |
| 4 | Foodweb descriptor | Cumulative distribution of biomass over TL: slope and position of inflection point. | Responds predictable to disturbance of community by fishing mortality. | EwE (North Sea, Baltic potential, Adriatic potential) |

| | | | | |
|---|---|---|---|---|
| 4.2.1 | Large fish (by weight) | LFI (proportion of biomass of individuals larger than a certain L threshold) | Responds predictable to disturbance of community by fishing mortality. | Gadget for modelled species, SMS similar (15-40 cm=small). Stocobar for cod. Size spectra model potential. All require decisions on lower cut-off and L |
| 4.2.1 | Large fish (by weight) | 95%-tiles of length of individuals in fish community. | Responds predictable to disturbance of community by fishing mortality. | Gadget for modelled species, SMS similar, Stocobar for cod. Size spectra model potential. All require decisions on lower cut-off. |
| 4.2.1 | Large fish (by weight) | LSI (proportion of biomass belonging to species larger than a certain Linf threshold) | Responds predictable to disturbance of community by fishing mortality. | Gadget for modelled species, SMS similar. Size spectra model potential. All require decisions on lower cut-off. |
| 4.3 | Abundance/distribution of key trophic groups/species | Total biomass of small fish | Information on food available to higher trophic levels in ecosystem. | Gadget, EwE for modelled species, SMS similar, Stocobar for cod. Size spectra model potential. All require decisions on lower and upper cut-off. |

| | | | | |
|---|---|---|---|---|
| 4.3 | Abundance/distribution of key trophic groups/species | Gini simpson diversity index for predators and prey | See above (1.7.1) | SMS for assessed species, EwE, Gadget potentially. |
| 4.3 | Abundance/distribution of key trophic groups/species | Gini-Simpson dietary diversity of each fish species. | See above (1.7.1) | SMS for assessed species, EwE, Gadget potentially. |
| 4.3 | Abundance/distribution of key trophic groups/species | Community biomass of pelagic, forage, demersal, benthos and total | Measure of where and how the biomass in the ecosystem is distributed | EwE all groups and Gadget, SMS and Stocobar for assessed species. |
| 4.3 | Abundance/distribution of key trophic groups/species | Average TL of community | See above (1.7.1) | See above (1.7.1) |
| 4.3 | Abundance/distribution of key trophic groups/species | Mean TL of the catch | See above (1.7.1) | See above (1.7.1) |
| 4.3.1 | Abundance trends of functionally important selected groups/species. [groups with fast turnover rates (e.g. phytoplankton, zooplankton, jellyfish, bivalve molluscs, short-living pelagic fish) that will respond quickly to ecosystem change and are useful as early warning indicators;<br>groups/species that are targeted by human activities or that are indirectly affected by them (in particular, bycatch and discards);<br>habitat-defining groups/species;<br>groups/species at the top of the foodweb;<br>long-distance anadromous and catadromous migrating species;<br>groups/species that are tightly linked to specific groups/species at another trophic level]. | | | Gadget, EwE, SMS and Stocobar for assessed species |

## 7.2 Methods for calculating indicators referred to in Table 7.1.1.

### 7.2.1 Loss in secondary production resulting from fishing. (L-Index)

The decrease in secondary production is proposed as a proxy for quantifying ecosystem effects of fishing, and it is formally defined in an index of ecosystem overfishing, the L (Loss in production) index (Libralato *et al*., 2008). The L-index is calculated by integrating the primary production required to sustain the catches (PPR) relative to the primary production (PP) in the ecosystem, the transfer efficiencies (TE, i.e. the efficiency in the transfer of energy from a trophic level to another) and the trophic level of the catches (TLc). Thus the formulation is based on properties of the catches (TLc, PPR) and of exploited ecosystems (PP and TE), allowing for estimation of the index from model outputs and directly from landings data. These input data are combined to measure the loss in secondary production due to fishing (the L-index) and to evaluate ecosystem effects of fishing. From catch data can, this be calculated as:

$$L = -\frac{1}{P \cdot 1\ T\ \mathrm{n}P} \cdot \sum_{i}^{m} \left( P_{E\ \ i} \cdot T\ \ {}^{T}P^{-1} \right) \equiv \frac{L}{E} - \frac{P_{i}}{R} \cdot T\ \ {}^{T}P^{-1}$$

Extension of this index work has provided each index value a probability of being sustainably fished (Psust) and, by fixing desired sustainability levels (e.g. 75% and 95%), it provides a basis for back-estimating the associated Ecosystem-based Maximum Sustainable Catches (EMSC; Libralato *et al*., 2008). The L-index quantification can be adapted to specific spatial scales (regional spatial assessment) and to large pelagic areas exploiting data from satellite for estimating PP, catches and available data on diets (for TL estimates).

Thus, the approach proposed integrates and complements previous analyses (Pauly and Christensen, 1995; Pauly *et al*., 1998; Tudela *et al*., 2005), allowing a broad and general application of the index using both landings data and ecosystem models. The L-index can give rough estimates of overfishing status and management advices measures, but it allows for elucidating regions of viable solutions (sensu Cury *et al*., 2005): within these solutions other constraints (community/population level considerations) can be defined and applied for the proper identification of best management options. The index might be useful, thus, used in combination with other approaches.

The loss in production index and the probability of sustainability of fishing have been used also to evaluate ecosystem overfishing at a global scale for the present and past decades using LMEs catch data (Coll *et al*., 2008). The index has also been applied to outputs of dynamic models of exploited ecosystems (Catalan Sea) allowing an evaluation of sustainability of fisheries along time for the past fishing history and for future scenarios of alternative management options (Libralato *et al*., 2005). Moreover, recently, the index was also related the effectiveness of fishing management to fishing sustainability (Mora *et al*., 2009).

### 7.2.2 Cumulative distribution of biomass over TLs

Accumulation of biomass has been documented for many marine foodwebs, with the middle TLs exhibiting the largest increase in cumulative biomass for a system (Gascuel *et al*., 2005, Link *et al*., 2009; Pranovi and Link, 2009). Changes to this accumula-

tion may reflect shifts in ecosystem structure and function, as well as represent important considerations for management thresholds. According to these observations, from a theoretical point of view, a perturbed ME should lower the stored, cumulative biomass and "stretch out" across TLs (Figure 7.2.1).

To describe and quantify these effects of the curve shape, the raw data are fitted according to a logistic non-linear regression model in order to estimate the slope factor/steepness of the curve, the inflection point in terms of TL and the Y-axis intercept (Figure 7.2.2).

A preliminary application of the method to 10 high-latitude ecosystems has been carried out within the context of the CAMEO workshop (Pranovi *et al.*, 2012).

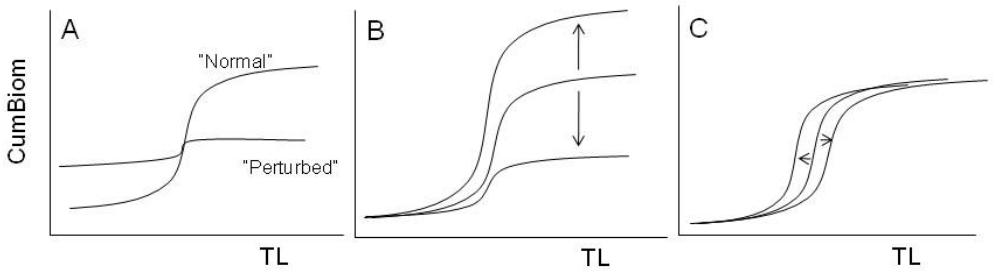


**Figure 7.2.1. Conceptual diagram illustrating (A) expected variations in the cumulative biomass–trophic level (cumB–TL) curve shape in relation to external drivers; (B) effects on the absolute curves; (C) effects on the relative curve.**

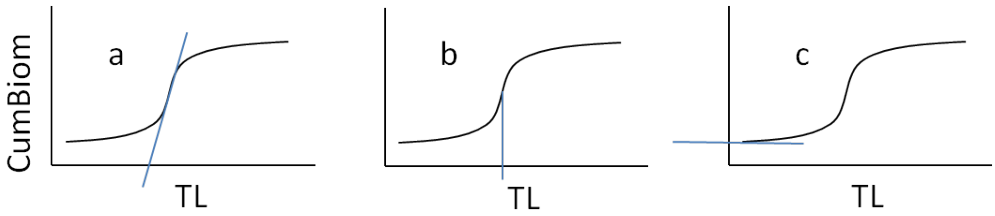


**Figure 7.2.2. Parameter used to describe the curve shape: a) slope factor/steepness of the curve; b) inflection point in terms of TL Y-axis intercept; c).**

### 7.2.3 Natural and total mortality

One of the most important features of multi species and ecosystem models is the quantification of natural mortalities (predation + residual mortality). Together with fishing mortality, the total mortality for different species and age (length) classes can be calculated. Fisheries management can use this information to ensure that total mortality is within suitable boundaries. From historic time-series it is possible to see whether current (or predicted) natural mortalities are inside historically observed ranges or whether the foodweb turns into a state not observed in previous years. Natural mortalities can be predicted by models. The information on natural mortalities is valuable for the interpretation of changes in stocks, as it is possible to disentangle fishing effects from foodweb related changes. The percentage of natural mortality in total mortality is thereby an easy to calculate indicator.

Changes in modelled natural and total mortalities can have different causes. For example, increasing natural mortalities can result from an increase in predator biomass but also from the depletion of alternative prey via the functional feeding response of

predators. Therefore, the indicator should only be interpreted in conjunction with additional foodweb indicators and information (e.g. large fish indicator, biomass of forage fish, predator abundance).

### 7.2.4 Mean weight of predators

Mean weight at age/ size can be related to the availability of food for predator populations in a certain area. Information on mean weight (at age or length) of key predators is available from surveys and commercial samplings. It is a standard input to assessments and trends are evaluated by routine. However, so far this is done for each stock separately. A comparison between species and stocks can give additional information on whether food becomes limiting in general or whether just some species or trophic guilds are impacted. Prediction of mean weight under different fisheries management scenarios is possible if significant relationships between weight of predators and the abundance/biomass of prey species in the model exist.

Changes in this indicator can be caused by changes in food availability as well as an increase or decrease in predator populations. The demand for food can be also influenced by temperature. Therefore, the indicator should be only interpreted in conjunction with additional information (e.g. biomass of forage fish, benthos, sea temperature, predator abundance, etc.).

### 7.2.5 Gini – Simpson index

In addition to information from community indicators such as LFI there is a need to take into account species diversity. It is inappropriate with GES to bring the foodweb into a state where only a few (large) predator or prey species dominate the system when the biomass of predators and prey was distributed more evenly in the system during the reference period. Just to look at the number of species is not enough as it often takes a long time to completely lose a species, while management should be informed and act much earlier. The Gini -Simpson index (1-D) applied to the predator and/or prey community provides the possibility to detect unwanted changes in diversity. Simpson's Diversity Index (D, Simpson 1949) is a measure of diversity which takes into account the number of species present, as well as the relative abundance of each species. As species richness and evenness increase, so diversity increases.

$$D = 1 - \left( \frac{\Sigma\, n(n-1)}{N(N-1)} \right)$$

**n = the total number of organisms of a particular species**

**N = the total number of organisms of all species**

### 7.2.6 Gini-simpson dietary index

Changes in the diet of predators can be often directly related to changes in the ecosystem. Predators can be seen as highly efficient "survey gear" detecting changes in the ecosystem potentially earlier than any fishing survey. If preferences for certain prey types are assumed to develop over a longer period, the diet of predators is more representative for the biodiversity in a given area then survey catches from one or two hauls randomly taken in space and time. A loss in the diversity of prey species found

in stomachs is therefore an early warning for changes in the ecosystem. The Gini - Simpson index (1-D) can be easily applied to diet information. The comparison between changes in the diet and in survey catches can provide additional valuable information.

## 7.3 Indicators selected for preliminary multispecies advice

Four community indicators were selected for presentation in the present version of multispecies advice (Section 11): Natural mortality by age, percentage of total mortality caused by natural sources, the Large Fish Indicator and the biomass by guild. None of these suggested foodweb indicators have associated reference points yet as reference points for the Large Fish Indicator pertains only to survey catches and not assessment output. Hence, time-series of the different indicators are given which can be supplemented by the development in indicators relative to agreed reference points as these are defined. Indicator development is given for approved key runs only (Baltic Sea SMS, North Sea SMS and North Sea Ecopath with Ecosim). The Barents Sea Gadget output was not included but can potentially be used to derive all four indicators.

In the North Sea, two indicators could be estimated using both SMS and EwE. The temporal development in the predictions from the two models is shown in figures 7.3.1–7.3.4. Note that inclusion of additional species in the EwE model grouping of 'Demersal' species reveals a different trend to when only the SMS demersal species are included.

At present, only the SMS output is shown in the multispecies advice in Section 11, since this model is more developed for providing the advice in appropriate format at present provides a longer time-series. Future work plans to investigate the possibility to incorporate more detail from the EwE model, particularly with regard to impacts on other species not included in SMS model and interactions among multiple fleets.

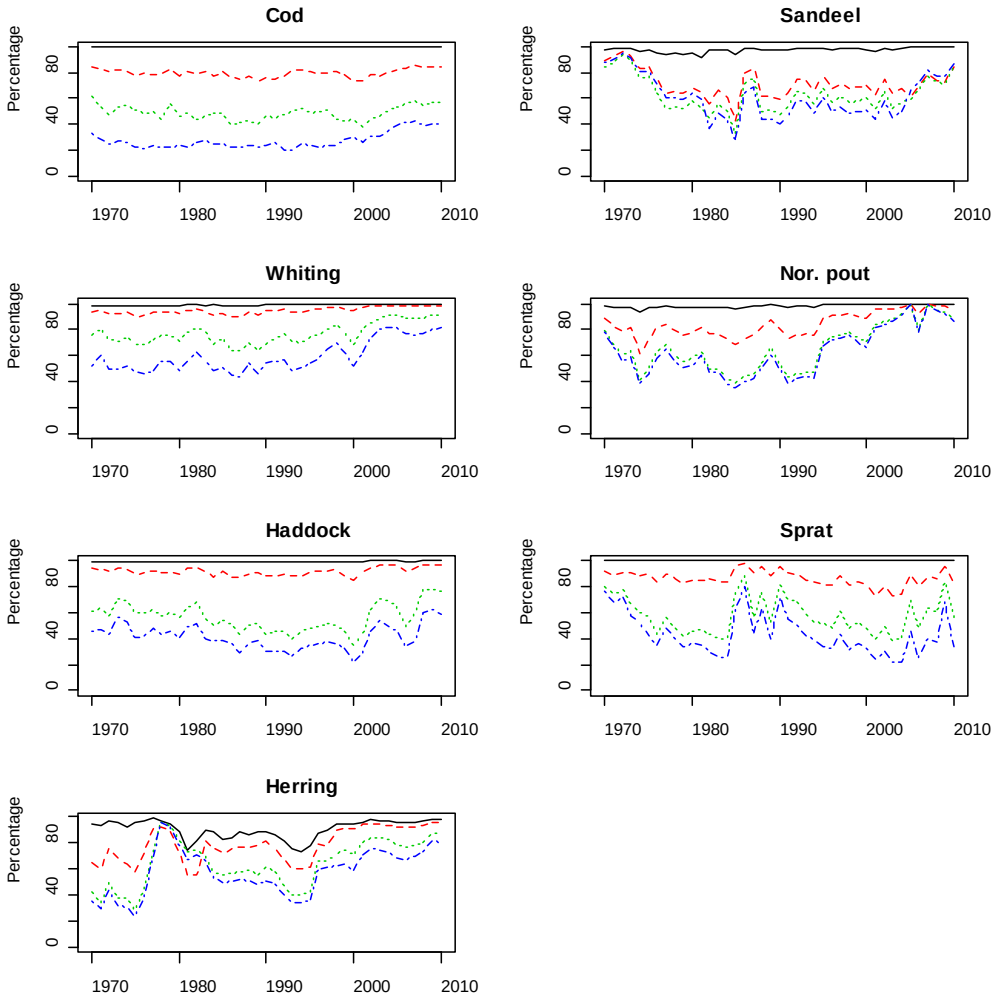


**Figure 7.3.1. Percentage of total mortality which is due to natural sources.**

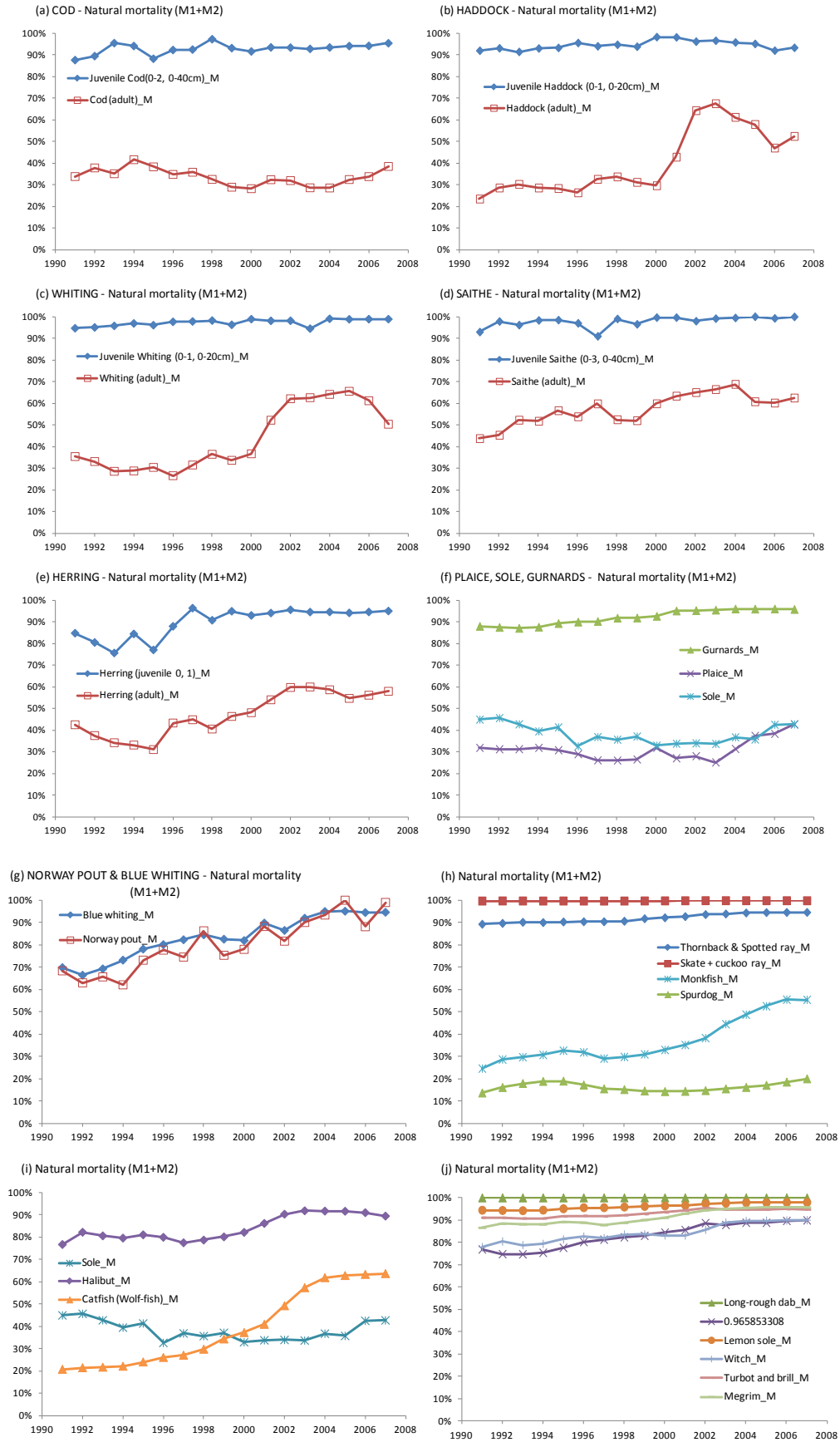


**Figure 7.3.2. Temporal development in natural mortality as estimated in the North Sea Ecopath with Ecosim.**

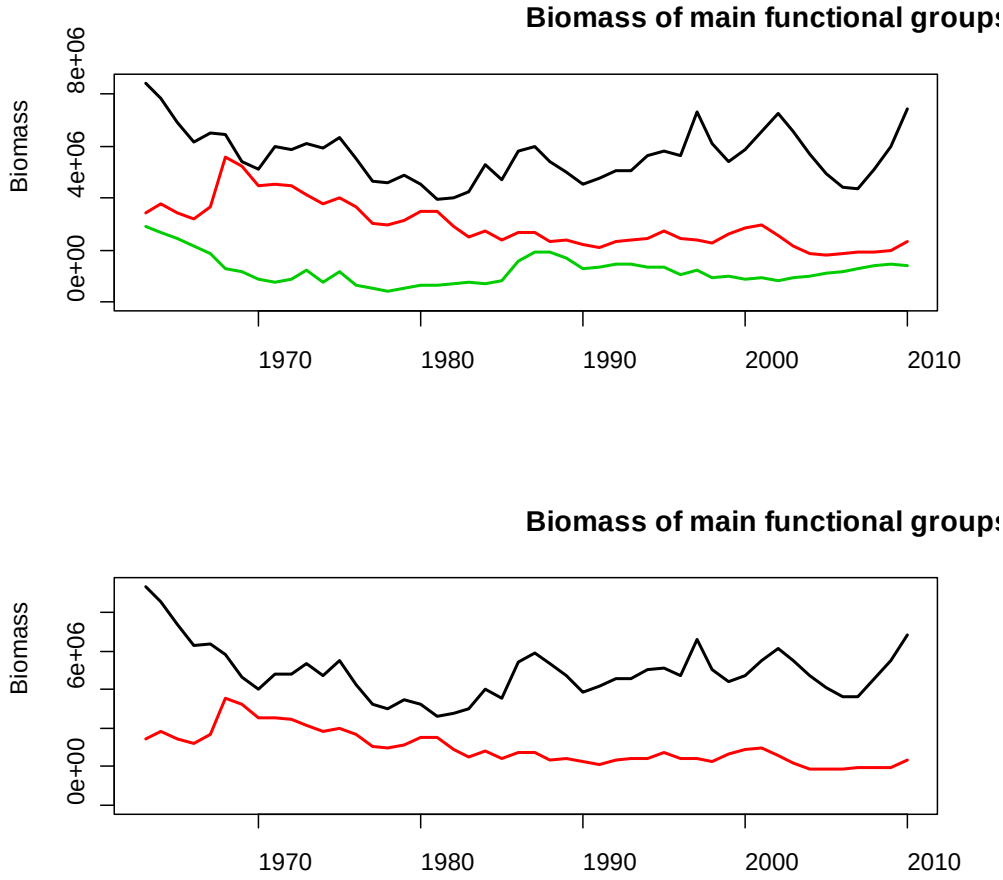


**Figure 7.3.3. Biomass of main functional groups (guilds). Upper figure: black = forage fish, red = demersal fish, green = pelagics (mackerel and horse mackerel). Lower figure: black = pelagics + forage fish, red = demersal fish (grey gurnard, starry ray, cod, whiting, haddock, saithe, plaice and sole).**

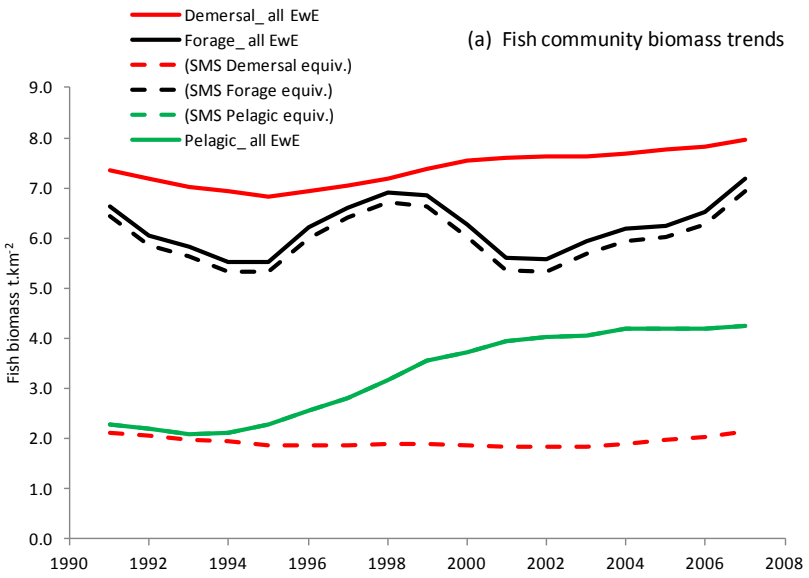


**Figure 7.3.4. Fish community biomass trends estimated from North Sea EwE and comparison with those estimated from SMS. black = forage fish, red = demersal fish, green = pelagics (mackerel and horse mackerel). Lower figure: black = pelagics + forage fish, red = demersal fish.**

## 7.4 References

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# 8 ToR f): Evaluate and explore size spectra models and compare to multispecies models in use in the ICES region

## 8.1 Request from WGECO on identifying foodwebs indicators

WGECO and WGFE (A. Rossberg) presented two requests to WGSAM for information via two short presentations.

The first was a request for the Jacobian approximation from WGSAM multispecies models. This gives a linearization of model behaviour, and is useful to analyse the stability properties of the foodweb model. The Jacobian can be used to:

a ) Compare the various models.

b ) Assist with analysis of MSY that the models would project.

c ) Determine which pressures would be associated with rapid system responses, and which with slow responses where system recovery from a pressure might be difficult or impossible.

d ) Aid with establishing relevant foodweb indicators and monitoring programmes.

e ) Predict size spectrum changes.

The second request was for information on how species richness, biomass and productivity would change if, for instance, 10% of the lowest size class or energy input to the system was removed. Would it drive changes in fish stock biomass that were >10%, 10%, or <10%? Obviously, the question if changes in primary production or producer abundance are amplified or damped as they propagate up the food chain is of fundamental importance.

Due to time constraints, these questions could not be investigated during the present meeting, but they should be considered in future.

## 8.2 LeMANS size structured model

Preliminary results from a variant of the LeMANS model (Hall *et al*., 2006) were presented; this was set up for the North Sea, with 21 fish species, divided into 32 5cm size classes. Predation was on the basis of relative lengths, and fish were recruited to the model at an assumed length of 2.5cm. Seven of the most important parameters were systematically varied within a range established by expert judgement, generating an ensemble of 28,800 different versions of the LeMANS model. Each version was run for 30 years under a scenario of assumed fishing at MSY. Outputs were then screened by reference to estimated stock abundances from ICES reports, and rejected if stock abundances at assumed MSY were below the lowest historic stock biomass.

Results were presented emphasizing the large fish indicator (LFI). The full range of parameters gave a simulated LFI of between 0 and 0.88, with 95% confidence limits of 0-0.59. The subset consistent with ICES stock estimates was much more constrained, giving a range from 0 to 0.17, with 95% confidence limits of 0.08-0.13. The reasons why the stock estimates are much more constraining than the parameter range estimates from experts remains to be investigated, though it appears that the stock–recruit relationship is a key source of uncertainty.

There were two major comments:

a ) It would be valuable to work back from the "validated" runs to see what this implied about the individual parameter ranges.

b ) Care should be taken when constraining the model with stock estimates, as these are at least partly dependent on single species stock assessment models that will be inconsistent with the LeMANS variants and with each other.

### 8.3 References

Hall, S. J., Collie, J. S., Duplisea, D. E., Jennings, S., Bravington, M., and Link, J. 2006. A length-based multi-species model for evaluating community responses to fishing. Can. J. Fish. Aquat. Sci., 63:1344–1359.

## 9 ToR g): Explore the trophic role of pelagic cephalopods (i.e. squid)

No progress was made on this ToR at WGSAM 2012 due to lack of attendance by members with this expertise. A more general ToR address similar issues was recommended for next year to afford some flexibility while retaining the main topical area.

## 10 ToR h): Explore the concept of Maximum Sustainable Yield (MSY) within a multispecies context and how it affects other biological reference points (BRP)

### 10.1 Balanced fishing as a possible approach to MSY

A number of recent papers (e.g. Garcia *et al*., 2012, Rochet *et al*., 2011) have promoted "balanced" fishing across all components of an ecosystem as an approach which could give a high yield with minimum disruption to the ecosystem. The possible approaches analysed range from constant selectivity across all species and size categories to targeting high productivity life stages rather than mature adult fish. In all cases the aim is to take a harvest such that impact is spread evenly over the ecosystem. This is in contrast to traditional fisheries management which has focused on protecting early life stages, and targeting large commercially valuable life stages. The proponents of the balanced approach have used an ecosystem analysis perspective. However if this approach were to actually be used in practice, then a fisheries approach would be required in to implement such a fishing strategy. Consequently, studies would be needed examining the short-medium term effects on the current fisheries and their target stocks.

Accordingly a study is being conducted to examine the likely impacts of such a strategy on the fisheries in the Barents Sea. This is an area which has highly profitably fisheries, essentially no subsidies, and the majority of stocks being considered to be sustainably managed (ICES CM 2012/ACOM:05). This success of traditional management, combined with a ban on discards, makes it an ideal area to compare current traditional management with a balanced fisheries approach. The study has three main areas of focus; examining how balanced fishing would interact with the species-specific biology of the main species; multispecies modelling of the short to medium term impacts on stocks and yields; and a consideration of the practicalities of implementing such a system.

### 10.1.1 Species biology

The specific biology for each species is critical for the fisheries and stock outcome under balanced fishing, and models which overlook this detail will not capture realistic effects. For example, Barents Sea capelin feeds for three to four years in the far north of the Barents Sea then migrates south to the coast of Northern Norway and Russia to spawn, and then dies. The spawning stock is highly variable between years (by up to two orders of magnitude), with good years producing far more larvae than can survive, and bad years being below the point at which recruitment is impaired. There is a small market for human consumption, but the majority are sold for fishmeal (so the oil-rich adult fish on their spawning migration are the most valuable per kg). For the last 15 years, practically all fishery on this stock has been carried out on mature fish prior to spawning. This biology has a number of implications for either balanced or productivity based fishing. First, setting a flat F between years results in a large loss of catch in years with good biomass, and adversely affects recruitment in poor years in return for a very minor catch. A second implication is that targeting the smaller fish requires much longer fishing trips, to catch much less valuable fish. It is doubtful that such a fishery would be economically viable, especially given the current high oil price. The capelin represents an extreme example, but the specific biology of each species will be important in evaluating the feasibility of any change in fishing patterns. Modelling studies that ignore such factors may thus be missing vital impacts.

### 10.1.2 Modelling

The multispecies modelling used a fisheries model rather than an ecosystem model, because any change in fishing must come through existing fisheries management structures. Fishermen are key stakeholders in modern fisheries management, this study thus focuses on the short to medium term effects on the current fisheries. The model therefore only examines the main interacting commercial species (cod, capelin, herring, minke whales and harp seals). The inclusion of marine mammals may be critical here as these typically target smaller species and size categories, and their exploitation of these needs to be considered if a model is to consider the effects of expanded human effort on these smaller fish. It thus does not account for the extra catch of currently non-commercial species. Nor does it account for catches at the lower trophic levels (plankton, benthos). These may add a large amount of biomass to the fisheries; however they are unlikely to add much profitability to the current fishing fleet. Targeting plankton may be profitable (and there is currently a small fishery on *Calanus finmarchicus*) but requires different gear to that employed by the current boats, and benthos fishery is likely to be neither feasible nor profitable with current gears and vessels. Results indicate that there is no single level of effort which produces similar stocks and catches across the different species. Further, for levels of effort which give similar stock sizes to those projected under current management, the yields are reduced under balanced harvesting. For all of these fish species, larger fish give higher price per kg, and fishing on the spawning migrations incurs the lowest cost. Consequently a switch from targeting large fish to catching mostly small fish would give an even higher reduction in profitability than the reduction in catches.

The model is run for 20 years. Thus any long-term effects (such as increasing SSB from low exploitation rates) are not considered. This is partly because we have no data to condition the model to these conditions, and thus the results must be highly dubious. For example the cod SSB is currently at its highest recorded level in the time-series going back to 1946.

Table 10.1.1 compares the average yield over twenty years between the basecase (currently agreed HCRs) and non-selective (balanced) fishing with different fishing mortality (F) values from 0.1 to 0.5. The highlighted values give the closest correspondence with predicted yields under current management, although it should be noted that a flat F=0.2 gives a long-term declining trend in the cod biomass. The highest biomass from a given yearclass for cod is typically around age 6, and fishing earlier reduces the overall yield. It should be noted that for all species a similar stock level to that predicted under agreed HCRs results in lower catches, and that it is unlikely that any of the species could reach levels much above their current ones (which are all at or near historic highs).

**Table 10.1.1 Predicted average annual stock size and catch under a variety of fishing scenarios. Basecase is the agreed size-selective HCR for each stock, fishing fraction sets a single F for all size categories of all species.**

| | | | | Fishing fraction | | |
|---|---|---|---|---|---|---|
| | **basecase** | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 |
| cod 3+ biomass | **2.27** | 4.62 | **2.23** | 1.71 | 1.3 | 1.1 |
| cod ssb | **1.26** | 3.55 | **1.46** | 1.01 | 0.73 | 0.56 |
| cod catch | **0.65** | 0.045 | **0.43** | 0.505 | 0.537 | 0.54 |
| | | | | | | |
| capelin biomass | **3.23** | 3.21 | **3.02** | 2.82 | 2.624 | 2.43 |
| capelin catch | **0.365** | 0.21 | **0.43** | 0.56 | 0.69 | 0.8 |
| | | | | | | |
| herring | **16.08** | 19.28 | 17.9 | 16.86 | 15.98 | 15.26 |
| herring juveniles | **3.23** | 2.96 | 2.66 | 2.41 | 2.2 | 2.02 |
| herring catch | **1.33** | 0.244 | 0.43 | 0.589 | 0.71 | 0.8 |

### 10.1.3 Implementation and enforcement

In terms of implementation, it is difficult to envisage any workable system. In contrast to a non-spatial model, fish do not distribute evenly over the ecosystem, not can one gear target all species. Balanced fishing would therefore require an extreme amount of micromanagement. Current management involves setting a quota for each species, a minimum catch size, a limit to how large a proportion of the catch can be undersized fish, a ban on discards and a range of management rules (such as gear restrictions and area closures) to enforce this. Fishers are then free to target whatever size category above the minimum catch size they choose. Under a balanced fishing regime all size categories need to be targeted equally, although not all are equally profitable. Thus quotas need to be set for each size category, and fishermen need to be penalized for both catching too much of a size category and too little of that category. This needs to be repeated for every species in the ecosystem, and if the balanced approach is to be taken at face value this means from plankton to whales. This obviously raises extreme demands on science to set these multiple quotas, and even more extreme challenges on management to ensure that the quotas are met for every size category of every species. An additional problem is that some of these species and/or size categories may be uneconomic to catch. In this case balanced harvesting would require government subsidies to ensure an even catch.

In a balanced fishing approach fishers would be forced to target species which are currently non-commercial. While it is possible in the long term to establish markets for fish which are not currently considered commercial, in the short term these are

likely to be sold at very low prices as industrial fish. The current balance of fleets and gears is tuned to match the current fishing practice. Changing to a balanced harvest would require major changes to the fleet, and would likely require substantial government support.

Using fishing by productivity rather than balanced fishing increases the demands on science, because the productivity at each size category needs to be known for every species, and potentially seasonal and annual variations should be considered.

There are also political challenges in gaining stakeholder acceptance for fishing at all levels of the ecosystem. In particular, catching and consuming animals from higher trophic levels (whales, seals, seabirds) may face opposition in some countries. Equally lower trophic levels (especially benthos) may face difficulties in finding a human consumption market.

### 10.1.4 Comparison to "Assembly fishing"

The balanced fishing approach may be contrasted to "assembly fishing" or "guild fishing" (see for example ICES, 2011). In this approach fish are grouped according to their life-history characteristics, and fishing is allowed to freely target fish within each guild. This is in contrast to balanced fishing, where all fish are essentially collected together regardless of their biology. The successful performance of assembly fishing in modelling simulations depends on appropriate grouping of species, and this is lost in balanced fishing.

### 10.1.5 Summary

In summary it is concluded that the balanced fishing approach would drastically reduce the profitability of the fisheries in the Barents Sea, and be essentially impossible to implement. Adopting a fishery based on the productivity of the life stages would somewhat improve the catches (though not necessarily the profitability) from the fisheries, but the regime would still be unimplementable. As a consequence such "balanced" fisheries studies may be considered an interesting academic exercise, but have little relevance to actual fisheries management. It also appears that even as an academic exercise it is critical to include realistic biology of the key species, as excluding this gives unrealistic results. It is also important that any comparison studies compare the outcomes of balanced fishing with that obtained by successful "traditional" management rather than against depleted stocks, and that biomass landed is not a meaningful comparison metric for commercial fisheries. It is also critical that marine mammals be included in the simulation, as these tend to target smaller fish, and excluding them gives a distorted picture of the total induced (natural and anthropogenic) mortalities.

## 10.2 Effort optimization to achieve $F_{MSY}$ in Ecopath with Ecosim

Many fishing fleets catch a range of commercial species and multiple fishing mortality targets exist for assessed species. Current $F_{msy}$ targets cannot be achieved simultaneously through changes in fishing effort alone. Recent work (Lynam and Mackinson in prep, Cefas) explores the objective solution to the problem of optimizing effort reductions in order to reach as close as possible to the fishing mortality targets for eight species of fish in the North Sea. To explore the effect of such a fishing strategy on the ecosystem, a foodweb model (EwE Key run, ICES, 2011) is projected forward, with and without climate forcing, and the direct fishing impact of the fleets and the indirect impacts propagated through the foodweb are evaluated. Many 'winners'

arise through reduced fishing, including cod and herring, but other species such as haddock, whiting and megrim 'lose', their biomasses declining as a result of predation by the ascending predatory species, which include cod, saithe and seals (Figure 10.2.1). Climate change has a strong effect on some groups (e.g. haddock) but little impact upon others (herring). The knock-on foodweb effects of climate change are important to consider in long-term management plans aiming to reconcile multiple objectives and faced with many trade-offs.

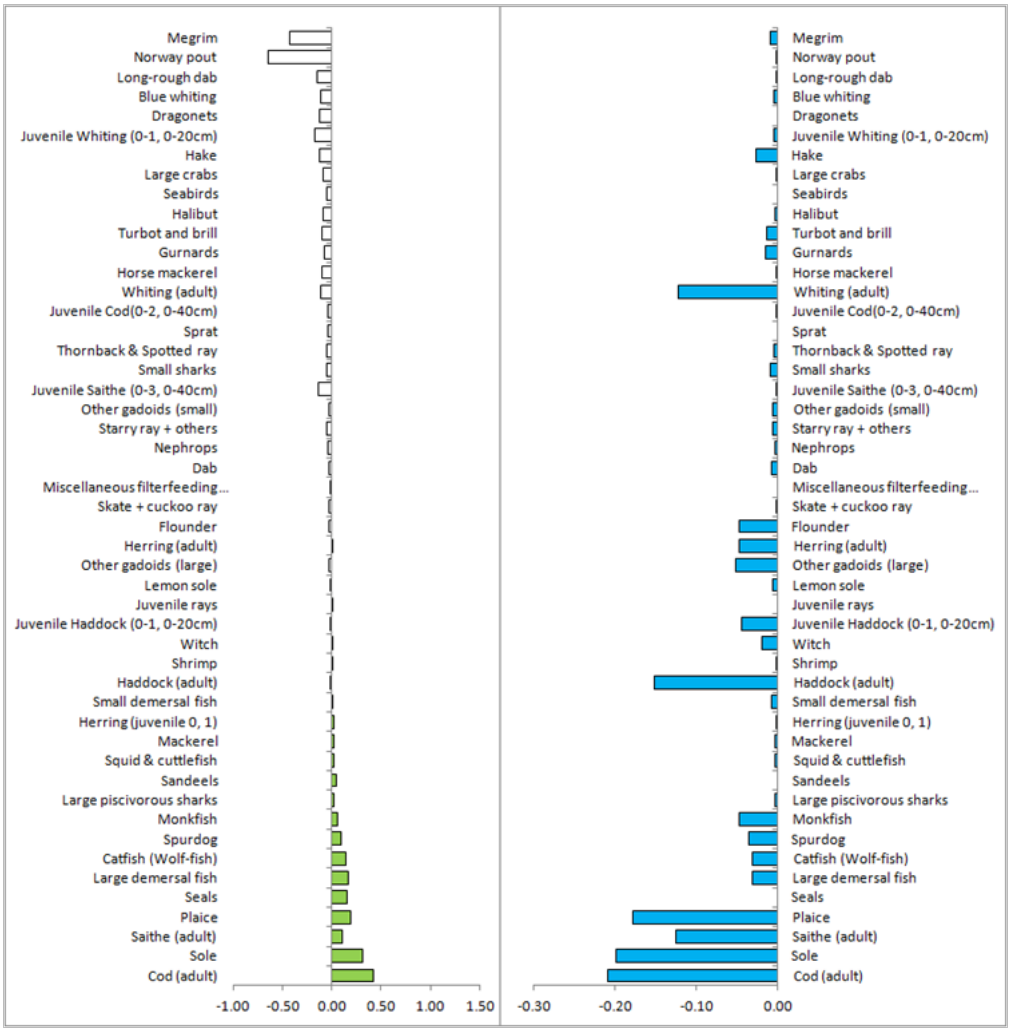


**Figure 10.2.1. Winners (green bars) and losers (white bars) in terms of percentage change in biomass in 2030 given percentage decreases in fishing mortality (blue bars) consistent with optimal fishing effort by fleet and including climate change effects.**

## 10.3 Cannibalism as an indicator for the MSY harvest control rule of cod in the Barents Sea

The NEA cod stock has been managed since 2004 under the precautionary harvest control rule, which is aimed at obtaining the MSY. In order to achieve the MSY, the HCR should provide the optimal correspondence between the stock size and carrying capacity from the long-term perspective. Cannibalism is an adaptive population mechanizm to control the cod abundance in the Barents Sea. It also enforces long-term correspondence between cod stock size and carrying capacity. Based on this we

can assume if the fishery management of cod in the Barents Sea is working well then the direction of year-to-year changes in fishing pressure should match the direction of change in cod cannibalism. Unlike the HCR the population mechanizm of abundance regulation has been tuned over a long period of time in response to ecosystem changes. Therefore we can consider the rate of cannibalism in the NEA cod as an indicator for a cod fisheries management aimed at achieving the MSY. Estimates of annual cod consumption by cod in the Barents Sea are available since 1984 and they may be derived from the AFWG reports (figure 10.3.1). According to the observations the cannibalism rate in the cod stock is sensitive to fluctuations in the capelin stock size in the Barents Sea. The negative link between cod cannibalism and capelin stock size may be disturbed and this is related to changes in cod stock size and its age composition, water temperature, and availability of alternative food (Bogstad *et al.,* 1994; Dolgov *et al.,* 2007; Yaragina *et. al.,* 2009).

The historical data show that the year-to-year changes in cod cannibalism and changes in cod catches were in the same direction in some years, but in other years were in the opposite direction (Figure 10.3.2). For example in 1986 both the cannibalism and catches increased compared with the previous year, but in 1998 they both decreased. However, in some years, as in 1989, cannibalism was increased while the catches, in contrast, decreased in comparison with previous year.

Over the period for which the cannibalism estimates are available, the various cod fishery management strategies were applied. Also, the recommended TAC did not always coincide with the landings. Therefore in order to evaluate how the precautionary HCR for cod in the Barents Sea will correspond to changes in cod cannibalism, the scenario modelling method was applied. The STOCOBAR model, which is able to simulate changes in NEA cod cannibalism as a function of capelin and cod stock sizes, and the abundance of young cod, was used (Filin, 2007).

5 simulations of the 100-year projections of cod stock development in the Barents Sea were performed. The precautionary management strategy was implemented. Uncertainties associated with TAC control were not taken into account. There was no restriction on how rapidly the TAC can change between years. The stochastic Ricker recruitment equation was used to couple the cod spawning-stock biomass and recruitment-at-age 1.

The temperature scenario for the long-term model runs was created by using historical data on annual water temperature at the Kola section for the period from 1951 to 2011. The cyclic climate variability was imitated by alternation of cold, moderate, and warm periods in the temperature scenarios. This was implemented by random selection of the historical replicates aggregated by cold, moderate, and warm years.

The capelin stock dynamics was simulated by using the assumption that the capelin stock size is dependent both on the cod stock size and its own size in the previous year. According to the historical data the probability of appearance of a large capelin stock is much higher if cod spawning stock in the previous year is lower than 400 thousand tons and if capelin stock in the previous year was more than 3 million tons. The following procedure was applied for producing the long-term stochastic projections of capelin stock in the Barents Sea. First, the historical replicates are drawn randomly from one of two datasets depending on the modelled cod spawning-stock biomass in the previous year (more or less than 400 thousand tons). At the second step the historical replicates are randomly selected from one of two datasets depending on the modelled capelin stock biomass in the previous year (more or less than 3 million tons). Then the arithmetic average for these two randomly selected values of

capelin stock biomass is calculated and this is put into the projection. This procedure is repeated for each modelled year.

Figure 10.3.3 demonstrates the simulated impact of capelin stock on the mortality rate of cod juveniles caused by cannibalism. According to the model, the relative numbers of cod consumed by cod substantially decreases as the capelin stock size increases. If the capelin stock biomass is larger than 3-4 million tons this link is enough strong, however if capelin stock is small this relation is weaker. When the capelin stock biomass is low, the range of cod mortality resulting from cannibalism is wide and this agrees with the observations.

Figure 10.3.4 shows the correspondence between the simulated year-to-year changes in cod consumption by cod and TAC. The position of the bars at the axes X indicates the changes in catches between years, while the size of the bars indicates the correspondent year-to-year changes in the annual consumption of cod by cod. The graph is split into3 segments, which are characterized by different states of the cod stock. The left segment reflects the depleted cod stock (annual decrease in catches more than 30%). The middle part of the graph reflects the cod stock in the relatively stable state (changes in annual catches in the range ±30%). The right segment in the graph reflects the cod stock in very good conditions (annual increase in catches more than 30%).

Figure 10.3.4 demonstrates that in these simulations the impact on cod stock from the fishery regulation and from cannibalism regulation may act in opposite directions. When fisheries pressure increases the cannibalism rate decreases, and vice versa. Especially strong differences between annual changes in cannibalism and TAC were revealed when the cod stock has grown rapidly (annual increase in TAC more than 50%). The explanation of these model outputs may be found in the multispecies context. If the cod stock grows rapidly it is means that the ecosystem conditions are favourable. Under favourable ecosystem conditions an abundant population can be supported. For cod this means increasing in rates of growth and maturation, as well as a decline in cannibalism and a reduced tendency to skip spawning. The STOCOBAR model captures this. In the model the size of capelin stock is a proxy for the cod carrying capacity. That is why we see in Figure 10.3.4 that when cod stock grows rapidly the mean size of capelin stock is larger than in periods when the cod stock is relatively stable or depleted. Therefore the simulated decline in cod cannibalism is a response to the growth in capelin stock size. This population response allows cod to get maximum benefits quickly from favourable changes in the ecosystems. The precautionary HCR for cod don't imply this consideration. For this reason in favourable ecosystem conditions annual changes in TAC will opposite direction compared with cannibalism. The too rapid growth of catches would make it difficult to make full use of opportunities for the growth of cod stock in the period when ecosystem conditions are favourable. This may have negative consequences for the MSY.

There are 2 ways to reduce the discrepancy between relative changes in cod TAC according to the precautionary HCR and cod consumption by cod. The simplest way is to use restriction on annual changes in TAC, which will restrain the otherwise too rapid increase of TAC in favourable ecosystem conditions. This management measure, with TAC changes restricted to +/-10% in each year, was introduced into the HCR for the Barents Sea in 2004. This restriction was introduced entirely for economic reasons. This modelling study shows that the TAC restriction is also reasonable from the ecosystem perspective. The existing limits in annual changes of the NEA cod TAC may be optimized by using this knowledge of cod cannibalism.

Another way to reduce the discrepancy between the relative changes in cod TAC according to the precautionary HCR and cod consumption by cod is to make the biological reference points for cod fishery management dependent on the capelin stock size or some other indicator of carrying capacity. This may be suggested as a management option.

### 10.3.1 References

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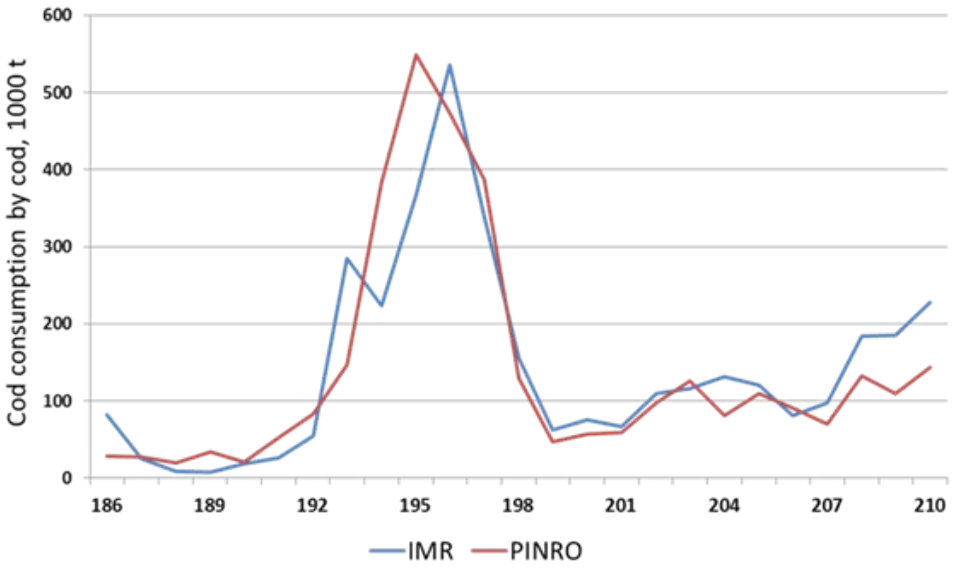


**Figure 10.3.1. Estimates of annual cod consumption by cod in the Barents Sea performed at the IMR and PINRO (AFWG, 2011).**

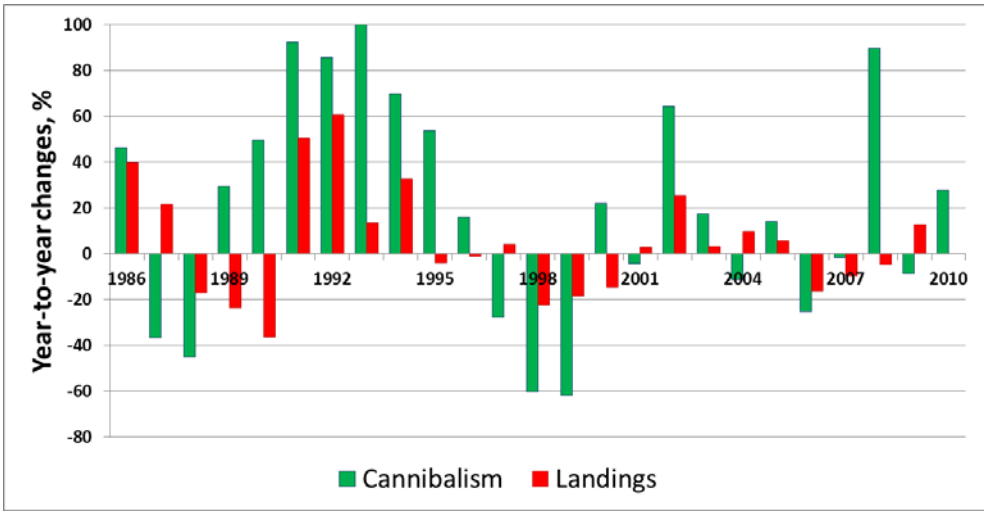


**Figure 10.3.2. Year-to-year changes in cod consumption by cod and catches (AFWG, 2011).**

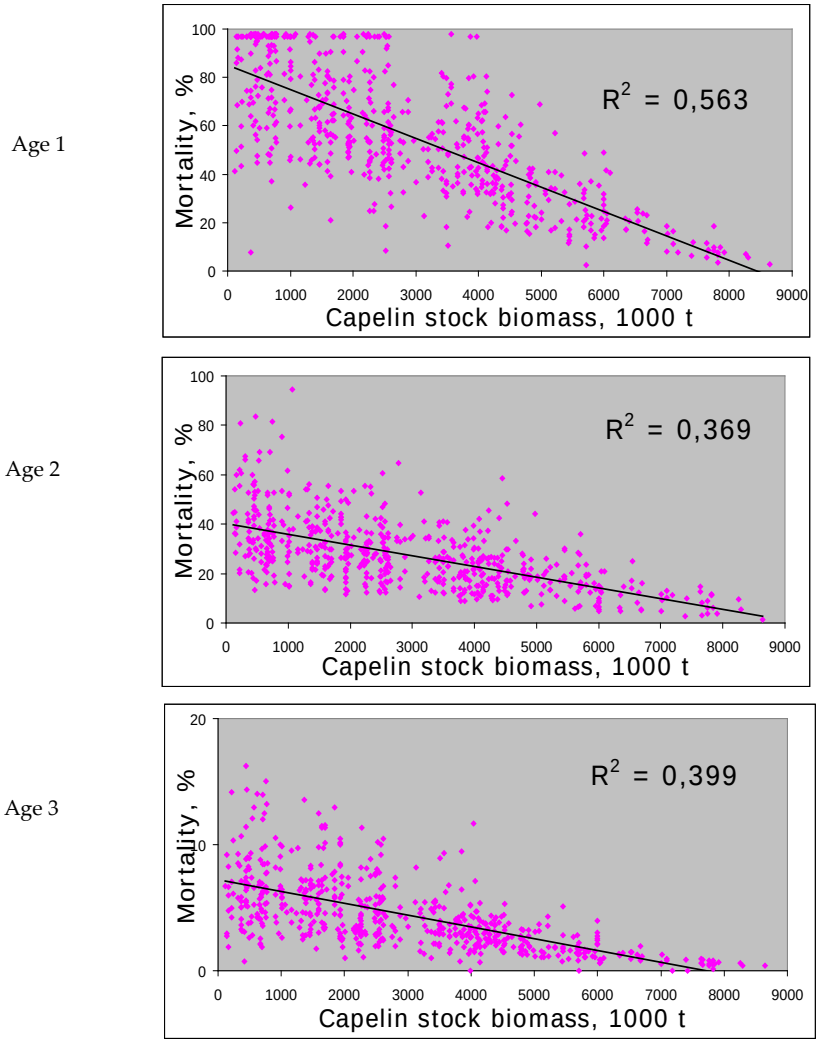


**Figure 10.3.3. Relationship between cod mortality rate due to cannibalism and capelin stock size (simulated data).**

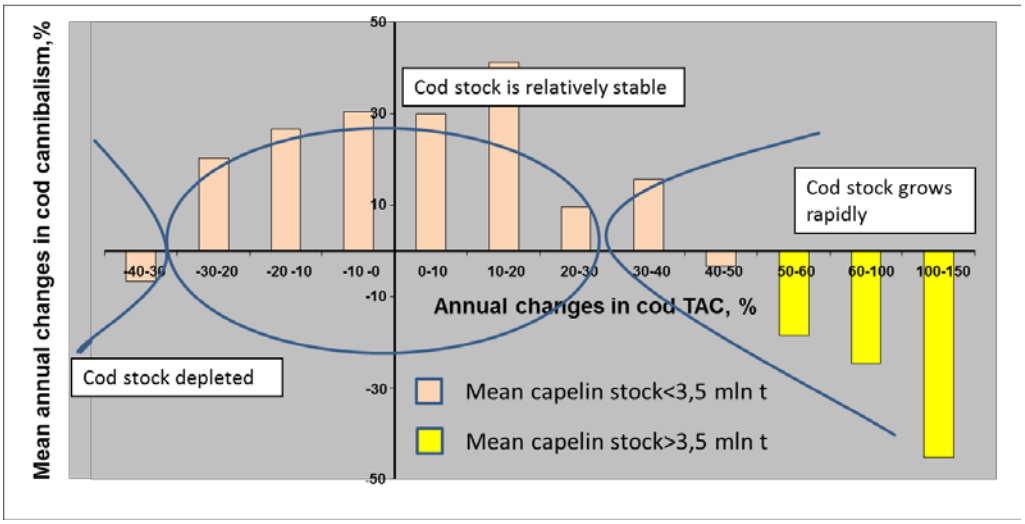


**Figure 10.3.4. Simulated year-to-year changes in cod consumption by cod against changes in cod TAC.**

## 10.4 Inherent limitations to predictability of the effects of exploitation

Ecosystem modelers often find that foodweb models are hard to calibrate. A frequently encountered problem is that, even when using the best available knowledge of biological and ecological parameters and interactions, models do not settle in at the observed system state, or, if a desired system state is enforced, it is not stable. Here we report on a simple analysis of a very simple, empirically parameterized fish-community model (WGSAM, 2011), which might explain the ecological mechanizm underling this uncertainty.

### 10.4.1 Model

The model and the fitting algorithm were previously described (WGSAM, 2011). In a recent, unpublished variant by Casper Berg, Anders Nielsen, Anna Rindorf and Martin Wæver, a Lotka-Volterra type model $dB_i/dt = (r_i - F_i - \sum_{ij} A_{ij}B_j)\ B_i$ accounts for changes the biomasses of 10 main species in the North Sea through time. The model parameters, given by the elements of the vector of linear growth rate $r_i$ and of the interaction matrix $A_{ij}$, were fitted to a 50-year long time-series for biomasses $B_i$ and exploitation rates $F_i$, which were generated by an SMS fit to all available survey data.

The response of equilibrium stock biomasses to changes in the exploitation rates is given by the inverse of the interaction matrix alone: $\Delta\mathbf{B} = -\mathbf{A}^{-1}\ \Delta\mathbf{F}$ (we use standard matrix-vector notation). Of particular interest is therefore the structure of the interaction matrix **A** and its inverse. The matrix **A** found by Berg *et al.* is, in units of $1/(10^6$ tons x year),

| | Cod | Had | Her | Nor | Pla | Sai | San | Sol | Spr | Whi |
|---|---|---|---|---|---|---|---|---|---|---|
| Cod | 0.047 | 0.0156 | -0.054 | -0.285 | -0.045 | 0.345 | 0.132 | 0.99 | -0.229 | 0.135 |
| Had | -0.492 | 0.4442 | 0.109 | -0.370 | 0.288 | 0.869 | -0.201 | -6.96 | -0.137 | 0.805 |
| Her | -0.122 | -0.2323 | 0.252 | 0.455 | -0.162 | -0.137 | -0.008 | -2.11 | 0.087 | 0.559 |
| Nor | -1.375 | -0.0027 | 0.274 | 1.744 | -1.022 | 0.522 | 0.068 | -9.99 | -0.882 | 2.033 |
| Pla | -0.094 | 0.1210 | -0.092 | -0.519 | 0.286 | 0.288 | 0.030 | 6.12 | 0.234 | 0.011 |
| Sai | -0.686 | -0.2833 | -0.020 | 0.046 | 0.273 | 1.052 | 0.072 | 0.46 | -0.138 | 0.441 |
| San | -1.605 | 0.0952 | -0.241 | -1.265 | 0.324 | 1.003 | 0.723 | 2.04 | 0.035 | 2.300 |
| Sol | 0.325 | 0.0871 | 0.087 | 0.322 | -0.764 | -0.067 | 0.035 | 7.65 | 0.310 | -0.906 |
| Spr | -0.179 | 0.2793 | -0.059 | -0.501 | 0.801 | -0.230 | 0.238 | 0.74 | 0.339 | -0.195 |
| Whi | -0.437 | 0.0314 | 0.019 | -0.467 | -0.792 | 0.474 | -0.105 | -2.43 | -0.474 | 1.858 |

### 10.4.2 Results and Discussion

The inverse of a matrix can be obtained by decomposing it into its eigenvalues $\lambda$ and left and right eigenvector, and replacing each eigenvalue by its reciprocal value $1/\lambda$. Components with small eigenvalues (i.e. where the absolute values are small) will therefore make a large contribution to **A**, and *vice versa*. Hence, it is important to understand the spectrum of the eigenvalue of **A** and its controlling mechanizms.

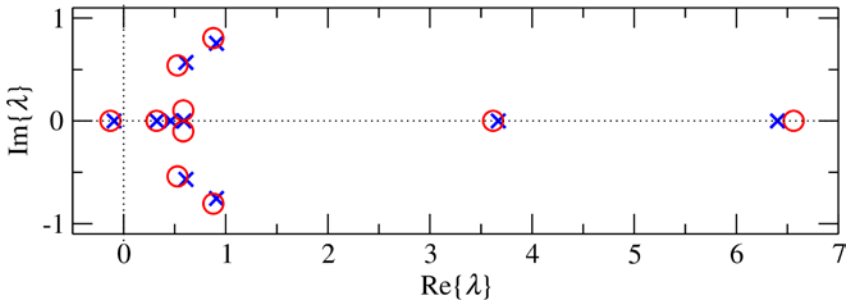


**Figure 10.4.1. Eigenvalues of matrix A above (blue x) in the complex plane and of the corresponding matrix after removing entries with values between -0.1 and 0.1 (red circles). Axes are drawn isometrically.**

The locations of the eigenvalues of the matrix **A** in the complex plane are shown in Figure 10.4.1 in blue. It can be seen that absolute values, i.e. the distances of the eigenvalues from the origin, vary substantially. For the eigenvalue closest to zero the corresponding eigenvectors make the largest contribution to $\mathbf{A}^{-1}$. Because the contribution of $\mathbf{A}^{-1}$ is determined as $1/\lambda$, small errors in the determination of this eigenvalue lead to large errors in $\mathbf{A}^{-1}$. We therefore also computed the eigenvalues for a modification of **A**, where all matrix entries with values between -0.1 and 0.1 were set to zero, similar to what would be done when removing statistically insignificant entries. The spectrum of the corresponding matrix is shown in Figure 10.4.1 in red. It can be seen that the distance by which eigenvalue shifted in the complex plane by this modification is of similar order of magnitude for all eigenvalues. However, while this shift has only small impact on the reciprocal of the eigenvalues near 6.5, it has a large impart on reciprocal of the eigenvalue closest to zero (it changes by about 25%). These observations are not specific to the case considered here; they reflect the general behavior of eigenvalue spectra under small perturbations of matrix entries. Even relatively small uncertainty in the estimation of the interaction coefficients $A_{ij}$ therefore leads to large uncertainty in the predicted response of the system to changes in exploitation rates, provided the matrix **A** has eigenvalues near zero.

Unfortunately, studies using community assembly models (e.g. WGECO 2012) show that, in the course of community assembly, interaction matrices naturally evolve to have a few eigenvalues near zero. The resulting structural instability is what constrains the number of species that an ecological community can accommodate. Communities without this feature can accept new species without losing existing ones, and therefore grow further.

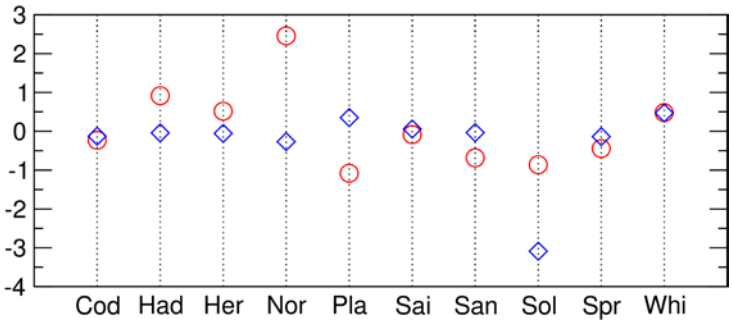


**Figure 10.4.2. Left eigenvector (blue diamonds) and right eigenvector (red circles) of A corresponding to the eigenvalue of that matrix which is closest to zero.**

Figure 10.4.2 shows, for the eigenvalue closest to zero in Figure 10.4.1, the left eigenvector of **A** (which gives the sensitivity of this component to changes in fishing pressure of each stock) and the right eigenvector of **A** (which gives the response of the stock biomasses to these pressures). Neither of the two eigenvectors is localized to a few stocks, which means that the uncertainty resulting from this small eigenvalue cannot be removed by lumping a few stocks as "unresolved" into a single model compartment. This, too, seems to be a general phenomenon: Eigenvalues near zero result from complex, diffuse interactions among many populations or stocks.

### 10.4.3 Conclusions

These observations suggest that there are large, inherent uncertainties in model predictions for the responses of natural ecological communities to external pressures. These uncertainties are the direct outcome of the assembly process structuring open communities. In the context of multispecies MSY, one needs to be mindful that a definition of a desired exploitation regime (MSY) in terms of a set of fishing mortalities rates ($F_{MSY}$) applied to stocks might be subject to these inherent uncertainties. The stock size obtained in reality might be different from those giving MSY. An idea to circumvents these problems has recently been proposed (Rossberg 2012) and is now under development.

### 10.4.4 References

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## 10.5 MSY and yield dependence of interacting species in the North Sea

### 10.5.1 Methods

Given the parameter estimates from the hindcast analysis (e.g. the 2011 key-run, WGSAM 2011), SMS can perform forecast scenarios. Exploitation pattern are assumed constant in the forecast period, but is scaled to a specified average F, or derived dynamically from Harvest Control Rules (HCR) and associated trigger values like $B_{lim}$. Recruits are produced from the applied stock/recruitment relation and an optional noise term.

#### 10.5.1.1 Optimization, maximizing total yield or value

Given the (fixed) parameter estimates from the hindcast analysis, SMS can estimate HCR parameter values (e.g. target F) given an objective function, e.g. total yield or as shown below total value.

$$f = \sum_{y=first\ year}^{last\ year} \sum_{s=1}^{no\ species} \sum_{a=0}^{last\ age} C_{y,s,a} * W_{y,s,a} * proportion\ landed_{s,a} * price_{s,a}$$

Two type of HCR can be specified:

1 ) Constant F ($F_{target}$)

2 ) F derived from the SSB in the start of the TAC year, and two SSB trigger values, $SSB_{low}$ and $BMSY_{trigger.}$ For SSB below $SSB_{low}$ F is set to zero and for SSB above $BMSY_{trigger}$ F is set to $F_{target}$. For SSB between $SSB_{low}$ and $BMSY_{trigger,}$ F is reduced linearly. The trigger values must be given as (fixed) input while $F_{target}$ is estimated by SMS.

The $F_{target}$ parameters are estimated within a specified range, e.g. 0 to Fpa.

Objectives function include yield weighted by a weighing factor given by species or by species and age. That gives the possibility to construct objective functions like:

- Total yield: sum of species yield
- Total value: sum of product of species yield at age and species price at age

The objective function can be calculated for one or more years. For deterministic recruitment a single year seems appropriate, e.g. year 2060 for a (near) equilibrium forecast.

#### 10.5.1.2 Stepwise definition of MSY

Reaching MSY of all species when a large number of species interact is not simple and may not even be possible if stochasticity in recruitment is included. Further, reaching MSY of some species may result in other stocks being below precautionary reference points in a large proportion of the simulated years. To avoid providing advice on the fishing mortality leading to MSY which is incompatible with the precautionary approach, combinations of target fishing mortalities which result in more than an agreed proportion of simulations being below e.g. $B_{lim}$ should be labelled as unprecautionary and should not be used to identify ranges compatible with MSY. Unfortunately, this requires a large number of simulations, and it was not possible to provide these during the meeting. Instead, a stepwise approach was taken to eliminate unprecautionary combinations of target fishing mortalities. This approach targets a balanced exploitation level for the various species, such that the individual species yield is determined by the species' life-history parameters rather than overall system maximization.

The method makes use of scenarios done for a large number of combinations of F values for the individual species. For a three species system, the approach written in pseudo code might look like:

```
For (CodF=0.3 to 0.6 by 0.05) {
  For (HerringF=0.2 to 0.4 by 0.05)
    For (SpratF=0.2 to 0.4 by 0.05)
        Make_scenario(CodF, HerringF, SpratF)
}
```

The results from the huge number of scenarios are presented in a way that illustrates the species yield as a function of the F values on the particular species itself and the F values on its predator and prey species as well. This output can then be used as basis for identifying ranges of F values that provide a sustainable fishery with the highest yield. A formal rule-based approach to identify a potential $F_{MSY}$ was tried during the WG, however the method appeared not fully developed. Instead the reduction of the range of F values to identify a potential $F_{MSY}$ was done was done in a qualitative *ad hoc* way.

#### 10.5.1.3 General settings of SMS hind- and forecast

The configuration of SMS hindcast was similar to the one used in the 2011 key run (WGSAM 2011). However, the stock recruitment relationships were changed for some species (see Section 10.5.1.4). The following predator and prey stocks were available: predators and prey (cod, whiting, haddock), prey only (herring, sprat, sandeel, Norway pout), predator only (saithe), no predator prey interactions (sole and plaice) and 'external predators' (8 seabirds, starry ray, grey gurnard, western mackerel, North Sea mackerel, North Sea horse-mackerel, western horse-mackerel, grey seals and harbour porpoise).

In forecast mode the population dynamics of all species except 'external predators' are estimated within the model from F values and model parameters. The populations of 'external predators' are kept constant.

Other values like mean weight at age are kept constant in the forecast. These values are based on the average of historical values.

- Mean weight in the sea: 2001–2010
- Proportion mature: 2001–2010
- Exploitation pattern: 2010. As SMS include a separable F model, the year range for exploitation pattern includes in reality more years
- Mean weight in the catch: 2001–2010
- Proportion landed: 2010 (to reflect the most recent gear development)
- Food consumption per individual: 2001–2010
- Stock size of external predators: 2006–2010.

Forecast scenarios were run with stochastic recruitment for the period 2011–2070. The mean values of the scenario output (SSB, yield, recruitment) for the period 2016–2070 were used as basis model output. By excluding the first 6 years the scenarios become reasonably independent of the initial stage of the stocks. With stochastic recruitment an equilibrium state is never obtained in the sense that stock size continue to vary between years.

#### 10.5.1.4 Ensuring precautionarity

The guidelines described under Section 11 were followed in this analysis as far as practically possible. Hence all F-values are estimated using harvest control rules for each species with zero fishing mortality when biomass is below $B_{lim}$ and a linear increase in F when biomass is between $B_{lim}$ and $B_{MSYtrigger}$ (defined here as $B_{pa}$) and a constant fishing mortality (target F) at biomasses above $B_{MSYtrigger}$. The approach requires biomass reference points for all species. For short-lived species (sandeel, Norway pout and sprat) which are managed using the "escapement strategy" (targets that SSB is above $B_{pa}$ after the fishery has taken place), total-stock biomass (TSB) was used as trigger point to reflect that the fishery is mainly focused one young and in some cases immature fish. The TSB reference points were derived from the SSB points raised by the ratio of long-term mean of TSB and SSB. Table 10.5.1 gives an overview of the applied reference points for the SMS analysis.

The scenarios do not take the assessment uncertainty into account, such that it is not possible to evaluate whether the HCR is precautionary in the traditionally used sense of the word which includes implementation uncertainty. Instead, we have chosen to evaluate precautionarity in two steps: 1) median SSB above $B_{lim}$ and 2) median SSB above $B_{pa}$.

#### 10.5.1.5 Stock recruitment relationships

Stock recruitment relations parameters are estimated in the hindcast SMS and their fit is included in the objective function as a penalty function. Limited work on selecting and fitting the SSB-R relations was done as part of the establishment of the key-run, as their actual values had a very limited influence on the overall fit. For forecast purposes SSB-R has a major influence and the previously used relations were revised. The revised SSB-R relations are shown in Figure 10.5.1. Compared to the previous key-run, "hockey-stick" with "known" inflection point relations have been substituted by Ricker relations where the maximum "Ricker" recruitment is within the observed range of SSB or the variance of the fit decreased.

### 10.5.2 Results

#### 10.5.2.1 System overview

The scenario SSB and yield are shown in figure 10.5.2 and 10.5.3. The SSB plot illustrates the direct and indirect responses to changes in F-values. As expected, the direct effect on an increasing F is a decreasing SSB (the diagonal of the figure). The indirect effect is however more complex. An increase in F on cod will give as smaller cod stock and less predation on its prey species. For whiting and haddock higher cod F leads to an increase in SSB (decrease in direct predation effect), while SSB of herring, sandeel, Norway pout and sprat will decrease (top row of figure 10.5.3). This decrease in SSB for the typically prey species is due to that the gain in whiting and haddock SSB induces more predation on the prey species than the predation from a larger cod stock. The effects of an increase in saithe F are similar to the effects of an increased cod F

There are also both direct and indirect predation effects of fishing on prey species. An increase in F on sandeel leads to a smaller sandeel biomass which induces a higher cannibalism for cod and whiting leading to a decrease in SSB for those species. This lower biomass predates less on haddock which there therefore increases with increasing F on sandeel, although sandeel is the main (fish) prey for haddock. The effects of an increase in F on sandeel for the other prey species herring, Norway pout and sprat

seems limited. Herring and Norway pout show a slight increase in SSB, probably because the decrease in cod and whiting biomass more than compensate the higher predation due to the reduction of sandeel.

The direct effect of F on yield (the diagonal of Figure 10.5.2) show a peak in yield for cod (F=0.45), haddock (F=0.3-0.4, range only precautionary if cod biomass is low) and sandeel (F=0.55) while the maximum yield for the other species is obtained at the outer range of F values presented. Whiting has the maximum yield at the highest F included (at 0.6) and saithe at the lowest F (at F=0.4). For herring, Norway pout and sprat the maximum yield is obtained at the highest F (at 0.55, 0.60 and 0.65 respectively). Given the selected range of F-values, the yield of all species except cod and saithe seems to be more affected by the indirect effect of fishing for other species (off diagonal figures) than by the direct effect of the individual species F (diagonal figures).

Recruitment seems to be rather insensitive to the F values (Figure 10.5.4) on the species itself, but some indirect effects, e.g. cod predation on whiting seems to give a substantial reduction in SSB and recruitment.

**Table 10.5.1. Reference point for the North Sea fish stocks as applied in the SMS analysis. Reference points in brackets are 'guesstimates', as the (single species) reference point does not exists or seems inappropriate in a multispecies context.**

| Stock | Blim | Bpa | MSY Btrigger (BESCAPEMENT) | mean F ages | Fpa | FMSY |
|---|---|---|---|---|---|---|
| Cod | 70 kt | 150 kt | 150 kt | 2-4 | 0.65 | 0.19 |
| Whiting | (200 kt) | (250 kt) | (250 kt) | 2-6 | (0.5) | |
| Haddock | 100 kt | 140 kt | 140 kt | 2-4 | 0.70 | 0.25–0.48 |
| Saithe | 106 kt | 200 kt | 200 kt | 3-6 | 0.40<br>(0.45) | 0.30 |
| Herring* | 800 kt | 1000 kt | 1300 kt | 2-6 | (0.35) | 0.24-0.30 |
| Sandeel | 430 kt | 600 kt | 600 kt | 1-2 | (0.5) | |
| Norway pout | 90 kt | 150 kt | 150 kt | 1-2 | (0.4) | |
| Sprat | (110 kt Bloss) | (150 kt) | (150 kt) | | (0.5) | |
| Sole | 25 kt | 35 kt | 35 kt | 2-6 | 0.40 | 0.22 |
| Plaice | 160 kt | 230 kt | 230 kt | 2-6 | 0.60 | 0.25 |

***revised Oct 2012.**

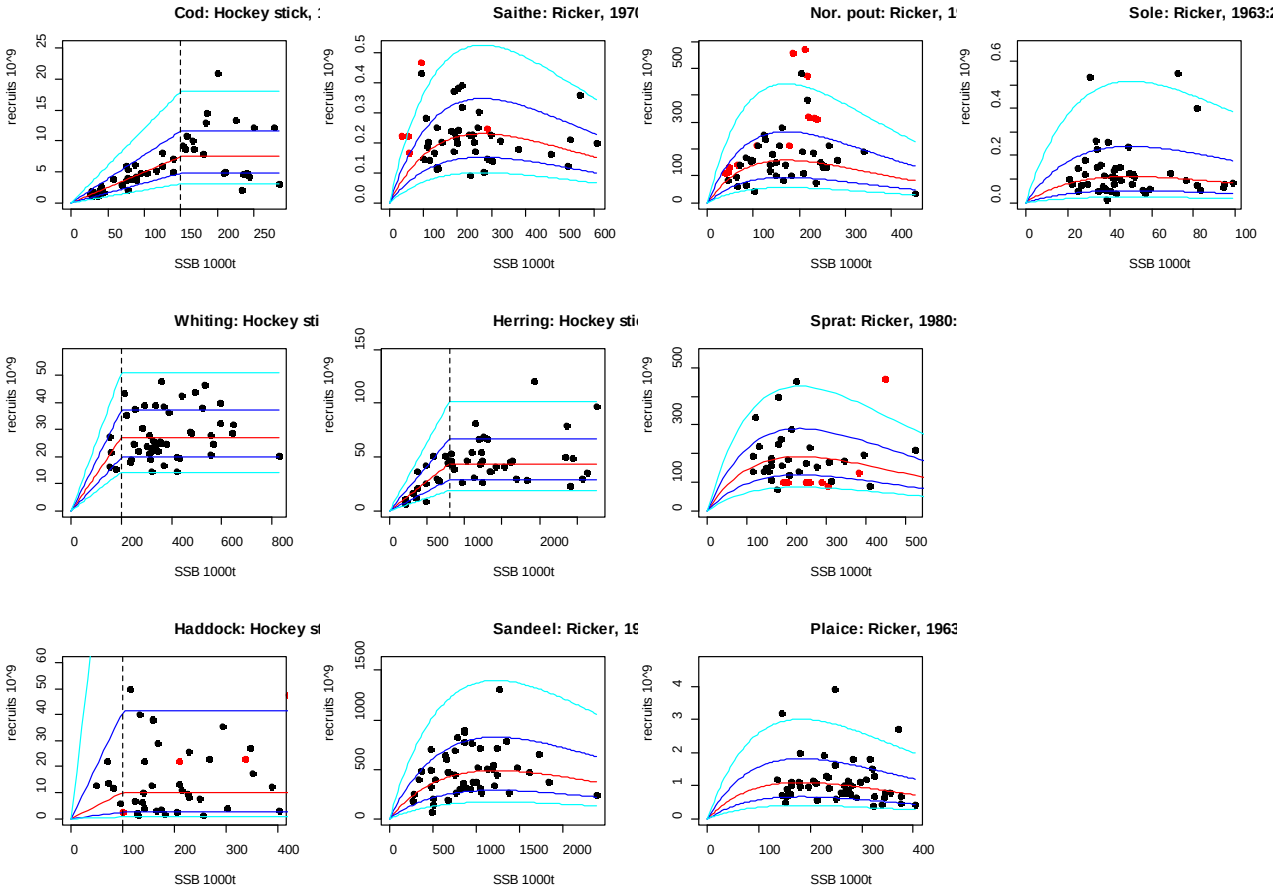



**Figure 10.5.1. Stock recruitment relations as used in SMS scenarios. The black dots are historical "observations" used in the fit. "Red" dots are not used in the fit. The red line represents the median and the blue and light-blue line the +- one and two standard deviation.**

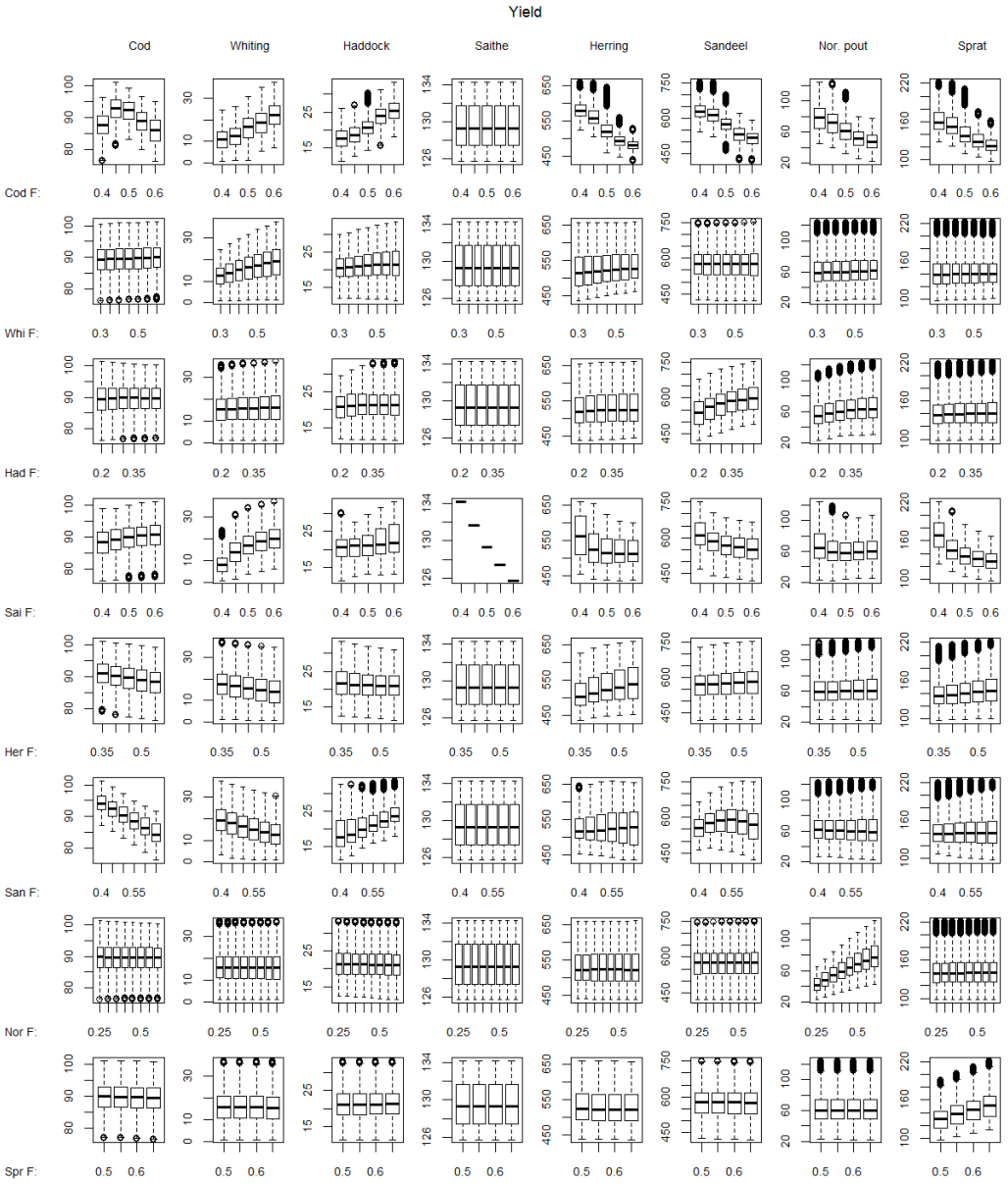



**Figure 10.5.2. Yield of (from left to right) cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat as a function of (top to bottom) target F on cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat. For guidelines on interpretation of the figure, see Section 11 on ToR i). Note that the F-values on the X-axis is the target F for SSB above the specified MSY B trigger biomass. For SSB below the trigger value, the F realized might be considerably lower.**

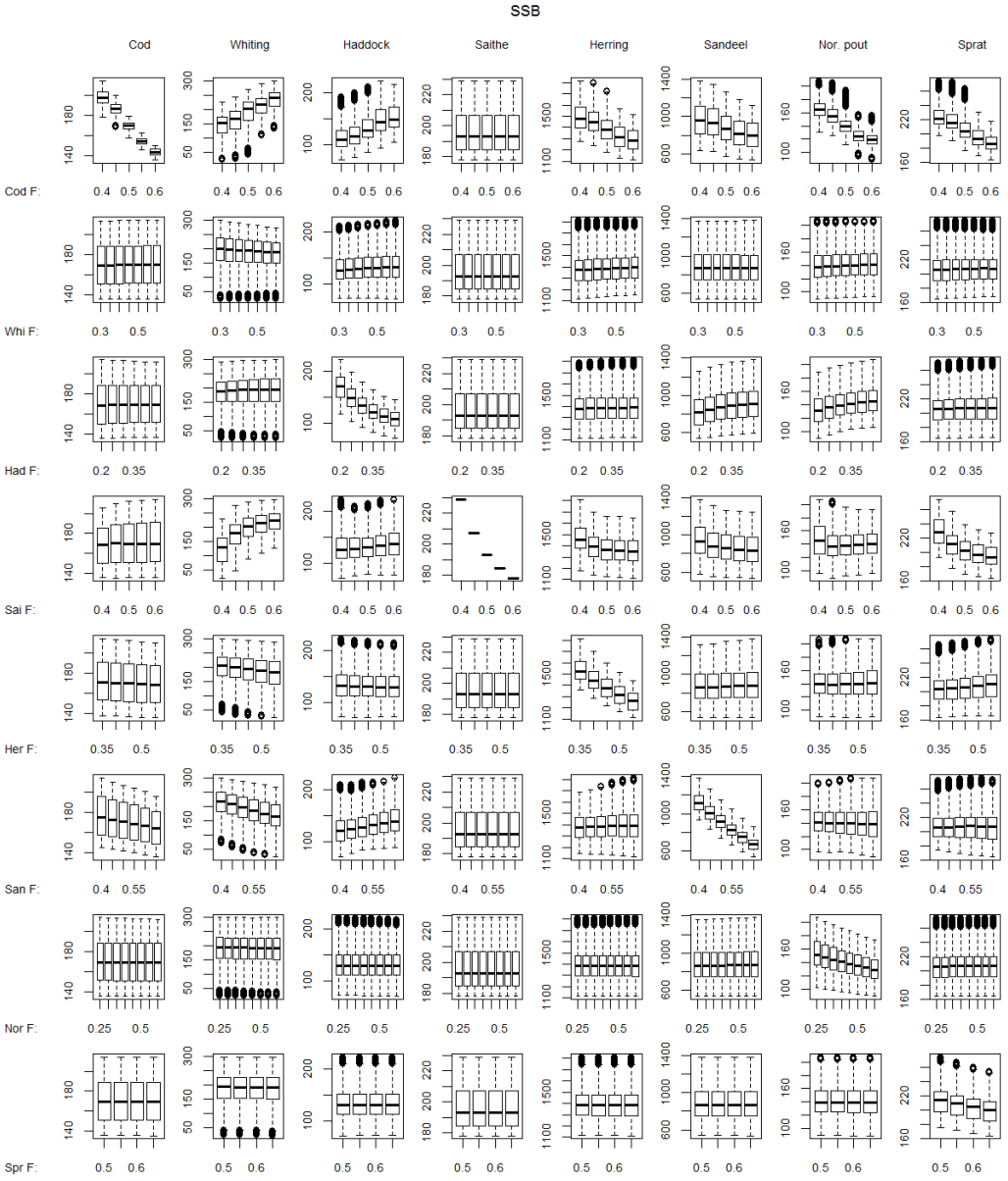



**Figure 10.5.3. SSB of (from left to right) cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat as a function of (top to bottom) target F on cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat. For guidelines on interpretation of the figure, see Section 11 on ToR i).**

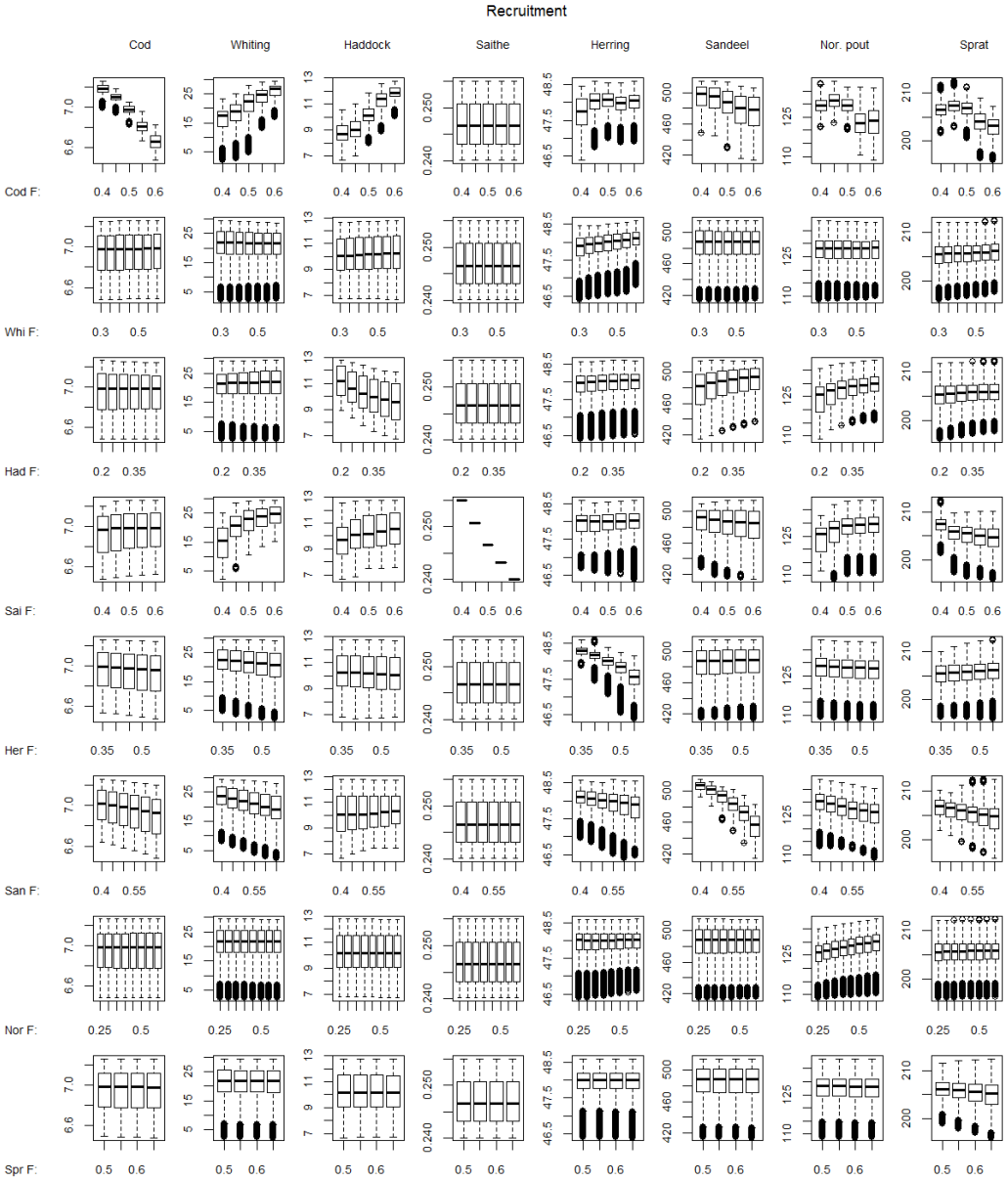



**Figure 10.5.4. Recruitment of (from left to right) cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat as a function of (top to bottom) target F on cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat. For guidelines on interpretation of the figure, see Section 11 on ToR i).**

### 10.5.3 Will fishing top predatory species (cod and saithe) at current single species estimate of $F_{MSY}$ allow prey stocks above precautionary biomass reference points?

Combinations of target F's where cod and saithe are fished at even moderately low F's result in cascading effects through the model as cod and saithe biomass increases resulting in increased natural mortality of haddock and whiting and subsequent lower biomass and yield of these species (Figures 10.5.2 and 10.5.3). Their prey (herring, sandeel, Norway pout and sprat) are then released of predation and increase substantially. Increasing the fishing pressure on cod and saithe leads to the reverse situation as cod and saithe biomass is decreased, haddock and whiting biomass increases and yield of herring, sandeel, Norway pout and sprat decrease. Because of these large effects of predation on haddock and whiting, it might not possible to maintain their

stocks above Blim when reducing fishing pressure on cod and saithe to their single species $F_{MSY}$.

Cod single species $F_{MSY}$ is currently 0.2 and 0.3 for saithe. Both $F_{MSY}$ are defined unconditional of SSB while the $F_{MSY}$ used in the SMS scenarios just applies for Biomass above Bpa. The lower range of F in the SMS simulations is 0.4 for the two species. Figure 10.5.5 show the SSB for combinations of F where F on cod and saithe are fixed to 0.4 and prey species SSB above Bpa. A low F-value (F<=0.25) maintain haddock SSB above Bpa (140 kt), but median whiting SSB is below 80 kt, and thereby considerably below Bpa (250kt) for combinations of F in the range 0.3-0.6. Due to time constraints, lower values of whiting F was not evaluated, however is seems unlikely that such values will provide a SSB three times higher than for whiting F=0.3.

In conclusion, fishing cod and saithe at F=0.4 and thereby at higher F than the single species $F_{MSY}$ might results in scenario SSB above Bpa and high catches for all species, except for whiting where SSB is considerably below Bpa.

The main predator on whiting is cod, saithe and grey gurnard. A larger stock of cod and saithe will result in a lower survival rate of whiting. However, whiting has previously had a rather large stock size concurrently with high stock sizes of cod and saithe. The main difference between the scenario situation and the state of the stocks in the 1980'ies is the assumption that the present high stock size of grey gurnard is maintained in the scenarios. Grey gurnard was responsible for more than half of the high M2 on 0-group whiting in the most present years whereas the M2 in the period before 1985 was quite low. The currently high stock size of grey gurnard might be a result of the present lack of large cod in the North Sea ecosystem: If a significantly larger cod stock will predate on grey gurnard, whiting SSB above Bpa might be possible for a situation with relatively large stock sizes of cod and saithe. Such large whiting stock will however reduce the prey stocks.

Mean F of cod has not been below 0.48 (in 1963) in the available time-series. A value of 0.4 is hence outside the range of historical which might introduce unreliable results.

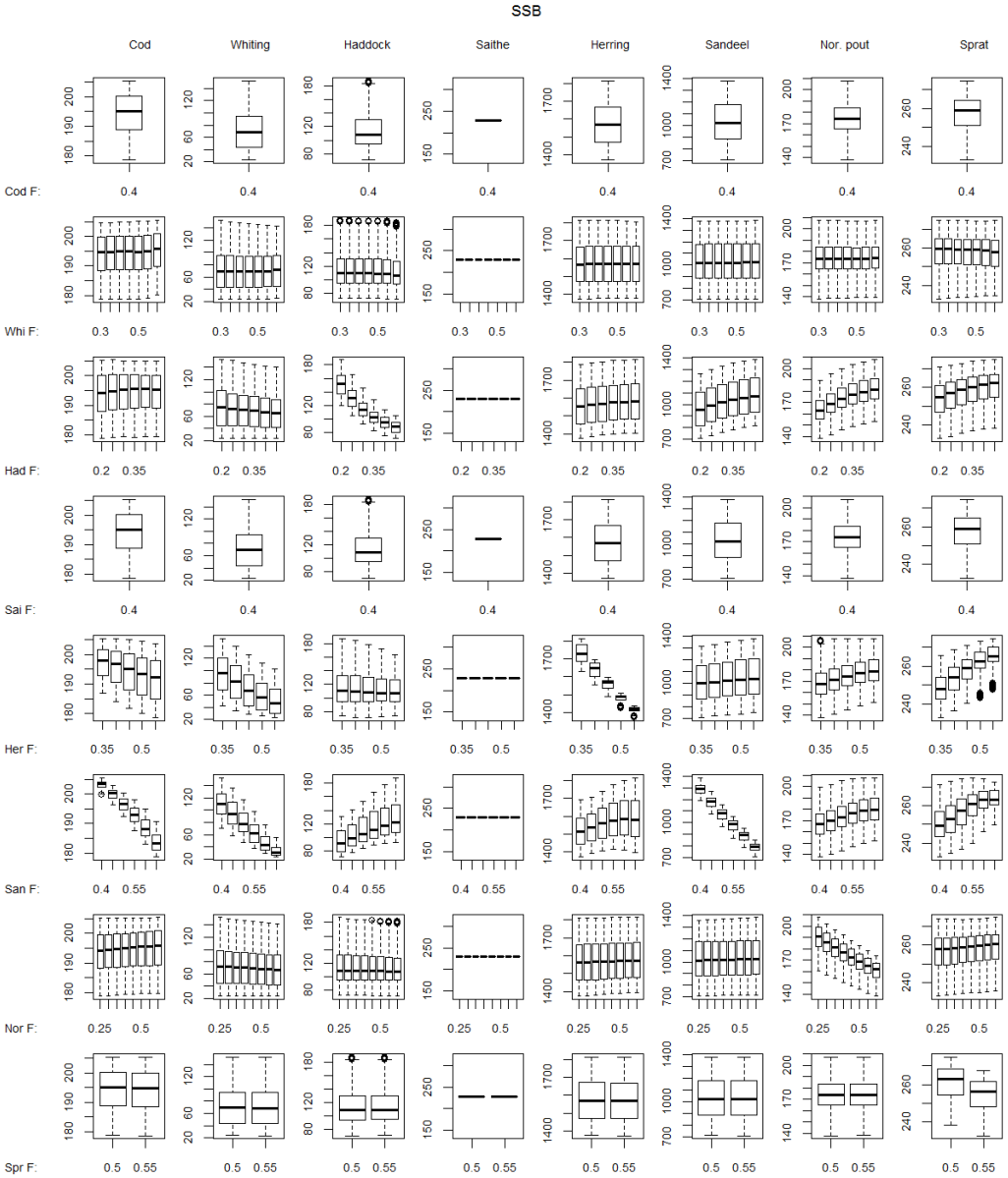



**Figure 10.5.5. SSB of (from left to right) cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat as a function of (top to bottom) target F on cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat. For guidelines on interpretation of the figure, see Section 11 on ToR i).**

### 10.5.4 Which species have yields which are greatly influenced by fishing mortality on other species and hence present important trade-offs?

Yield of virtually all species are strongly affected by the target F on cod and saithe and the effect of changes in predation on yield is much as large or larger than the effect of target F on all species. A high F on cod and saithe releases the two smaller gadoids from predation and increases both biomass and yield of them. For haddock yield remains virtually unrelated to haddock F (Figure 10.5.2). However, for the remaining species, there is a reasonable relationship between species-specific F and yield. All species have at least two strong off-diagonal interactions, and whiting and sandeel yield are strongly affected by F on three other species (saithe, cod and sandeel and saithe, cod and haddock, respectively). Norway pout and sprat are the only two species where F does not significantly affect yield of other species. Whiting

and haddock fishing mortalities are the second least important for other species with only two interactions each.

**Table 10.5.2. Importance of fishing mortality (rows) on yield (columns). Dark shading indicates high importance, light indicates low importance. Criteria: Dark: Median, upper and lower whiskers change, medium grey: median, upper and lower whiskers change but change is small, light grey: upper or lower whiskers change, white: not noticeable change in median or whiskers. White line indicates effects of species F on species yield.**

| F\Yield | Cod | Whiting | Haddock | Herring | Sandeel | Nor. pout | Sprat |
|---|---|---|---|---|---|---|---|
| Saithe | | | | | | | |
| Cod | | | | | | | |
| Whiting | | | | | | | |
| Haddock | | | | | | | |
| Herring | | | | | | | |
| Sandeel | | | | | | | |
| Nor. Pout | | | | | | | |
| Sprat | | | | | | | |

### 10.5.5 Which combinations of fishing mortality will lead to biomasses of all modelled species above Blim?

With target F at 0.4 on both saithe and cod, SSB of whiting is less than 80 kt (Figure 10.5.3) and thus far below any candidate for Blim. Maximum yield of cod is obtained for F in the range 0.45 to 0.50 (Figure 10.5.2) with median SSB above Bpa (Figure 10.5.3). Cod F at 0.5 seems therefor to be within precautionary limits for cod and to allow the whiting stock to increase. Saithe is another important predator on whiting (Figure 10.5.3). Saithe yield is highest (134 kt) for F=0.4, but F=0.45 gives practically the same yield (132 kt) with a median saithe SSB around 20kt less than for target F=0.4. Median saithe SSB for F=0.45 is slightly above Bpa (200 kt) such that F=0.45 seems precautionary. Figure 10.5.6 present SSB for cod F>=0.5 and saithe F>=0.45. Median SSB is above Blim (Table 10.5.3) for the given F level at the particular species. This median includes combination of F for other species, such that the minimum SSB for a particular F combination might be lower than Blim (Table 10.5.3 lower part) All the statistics are based on the median SSB over the years 2016-2070, which means that SSB in individual years might be below $B_{lim}$. More work is needed to provide improved performance statistics.

**Table 10.5.3. SSB for combinations of F with cod F>=0.5 and saithe F>=0.45.**

Median SSB (1000 tonnes)

| | 0.2 | 0.25 | 0.3 | 0.35 | 0.4 | 0.45 | 0.5 | 0.55 | 0.6 | 0.65 |
|---|---|---|---|---|---|---|---|---|---|---|
| Cod | NA | NA | NA | NA | NA | NA | 170 | 154 | 144 | NA |
| Whiting | NA | NA | 240 | 236 | 233 | 230 | 227 | 224 | 222 | NA |
| Haddock | 187 | 165 | 148 | 136 | 127 | 121 | NA | NA | NA | NA |
| Saithe | NA | NA | NA | NA | NA | 207 | 193 | 184 | 178 | NA |
| Herring | NA | NA | NA | 1455 | 1372 | 1300 | 1236 | 1182 | NA | NA |
| Sandeel | NA | NA | NA | NA | 1045 | 943 | 852 | 770 | 698 | 619 |
| Nor. pout | NA | 138 | 135 | 131 | 128 | 126 | 123 | 121 | 119 | NA |
| Sprat | NA | NA | NA | NA | NA | NA | 197 | 193 | NA | NA |

Minimum SSB (1000 tonnes)

| | 0.2 | 0.25 | 0.3 | 0.35 | 0.4 | 0.45 | 0.5 | 0.55 | 0.6 | 0.65 |
|---|---|---|---|---|---|---|---|---|---|---|
| Cod | NA | NA | NA | NA | NA | NA | 161 | 146 | 136 | NA |
| Whiting | NA | NA | 126 | 126 | 125 | 124 | 123 | 126 | 130 | NA |
| Haddock | 136 | 118 | 109 | 103 | 97 | 94 | NA | NA | NA | NA |
| Saithe | NA | NA | NA | NA | NA | 207 | 193 | 184 | 178 | NA |
| Herring | NA | NA | NA | 1362 | 1286 | 1223 | 1168 | 1117 | NA | NA |
| Sandeel | NA | NA | NA | NA | 935 | 839 | 741 | 666 | 599 | 540 |
| Nor. pout | NA | 102 | 100 | 98 | 97 | 95 | 93 | 91 | 89 | NA |
| Sprat | NA | NA | NA | NA | NA | NA | 176 | 171 | NA | NA |

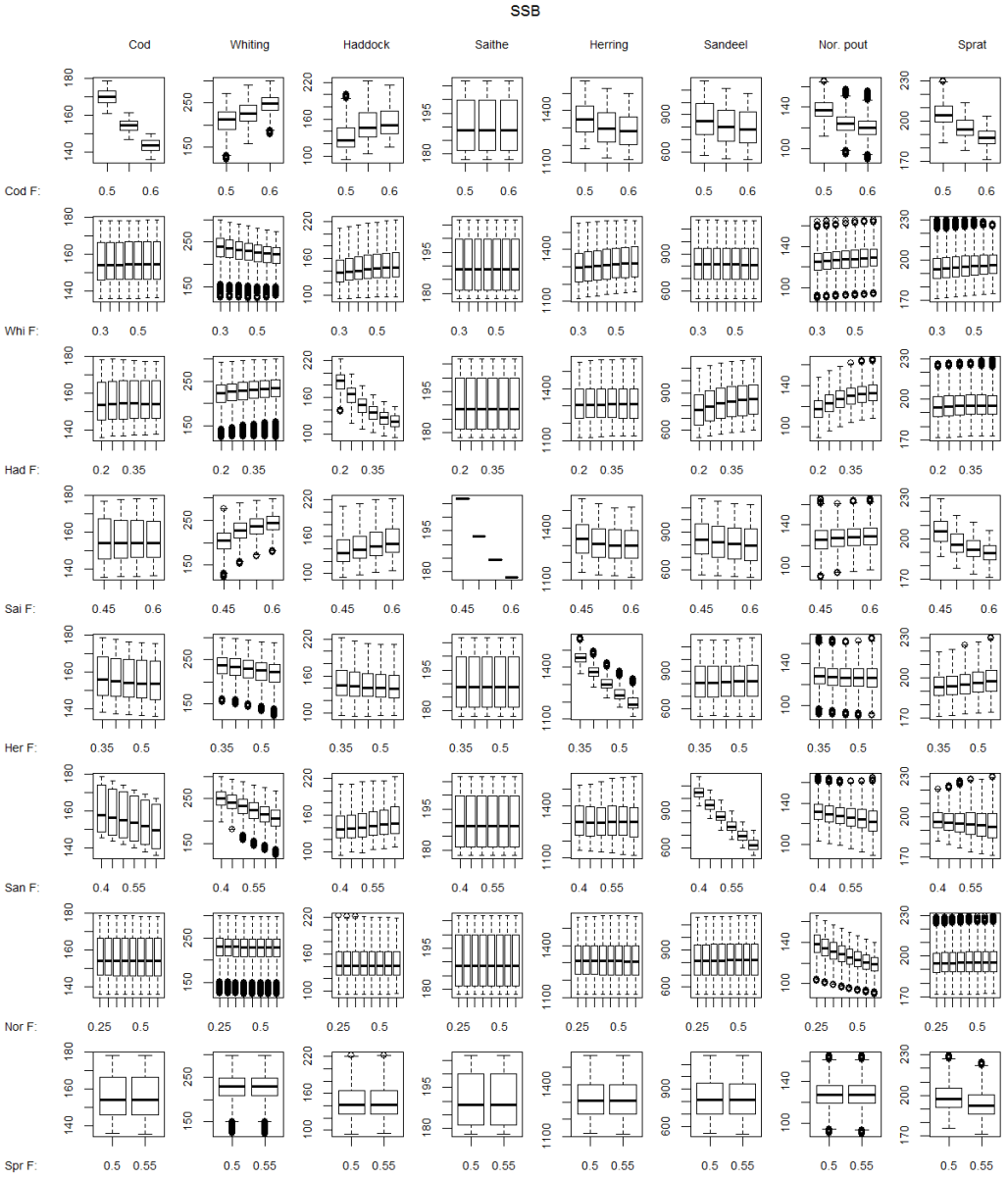



**Figure 10.5.6. SSB for cod F>=0.5 and saithe F>=0.45.**

### 10.5.6 Maximizing total yield or value

For deterministic recruitment, the identification of candidates for MSY can be estimated from maximization of yield or value as outlined in Section 10.5.1.

The highest value of the landings, without any constraints on biomasses, is obtained by depletion of the whiting and haddock stock (Figure 10.5.7), such that their SSB become clearly below Blim.

By using a penalty function for SSB below Blim, it is possible to maximize total value, given all SSB above Blim (Figure 10.5.8). Target F for the predators are then estimated to the maximum value of range [0; Fpa] used to bound the Ftarget parameters. SSB does not reach Bpa in all cases and the realized F becomes lower than Fpa because of the implemented harvest control rule. SSB of the prey species herring, sandeel and sprat becomes higher than $B_{pa}$ but yield is below the long-term average of historical yield. With an unconstrained Ftarget parameter, Ftarget (and realized F) becomes

higher (Figure 10.5.8) for the all the predators and SSB lower for the predators except haddock. Yield and SSB of herring and Norway pout increases with the higher F on the predators, while the values for sandeel and sprat are more or less the same.

Maximizing total value with the constraint of having SSB above Bpa results slightly higher SSB for the predators and most of the prey (Figure 10.5.9) compared to the maximization with Blim as the constraint (Figure 10.5.8), however F and yield become lower.

Maximizing total yield (weight; Figure 10.5.10) with the constraints SSB > Blim gives similar result as maximizing total value (Figure 10.5.9).

The optimization scenarios show that the result (target F) depends very much on the objectives (objective function and SSB constraints). It is possible to maintain all species SSB above Blim and maintain a fishery with significant yield and value. The exact combination of species target F depends however on the exact definition of the objectives, and the weighting factors (e.g. price pr kg) actually used for calculating these objectives. This makes the approach less robust for use in management.

All the optimizations were done with deterministic recruitment which leads to an equilibrium state and thereby facilitate the minimization of the objective function. The dynamics of the system however become very different from a real system (simulated with stochastic recruitment). Runs with stochastic recruitment using and cumulated yield or values over e.g. 100 years as objective function were also tried, but the results were depending on the number years actually used in the simulation and the random number seed used for generating the stochastic recruitment.

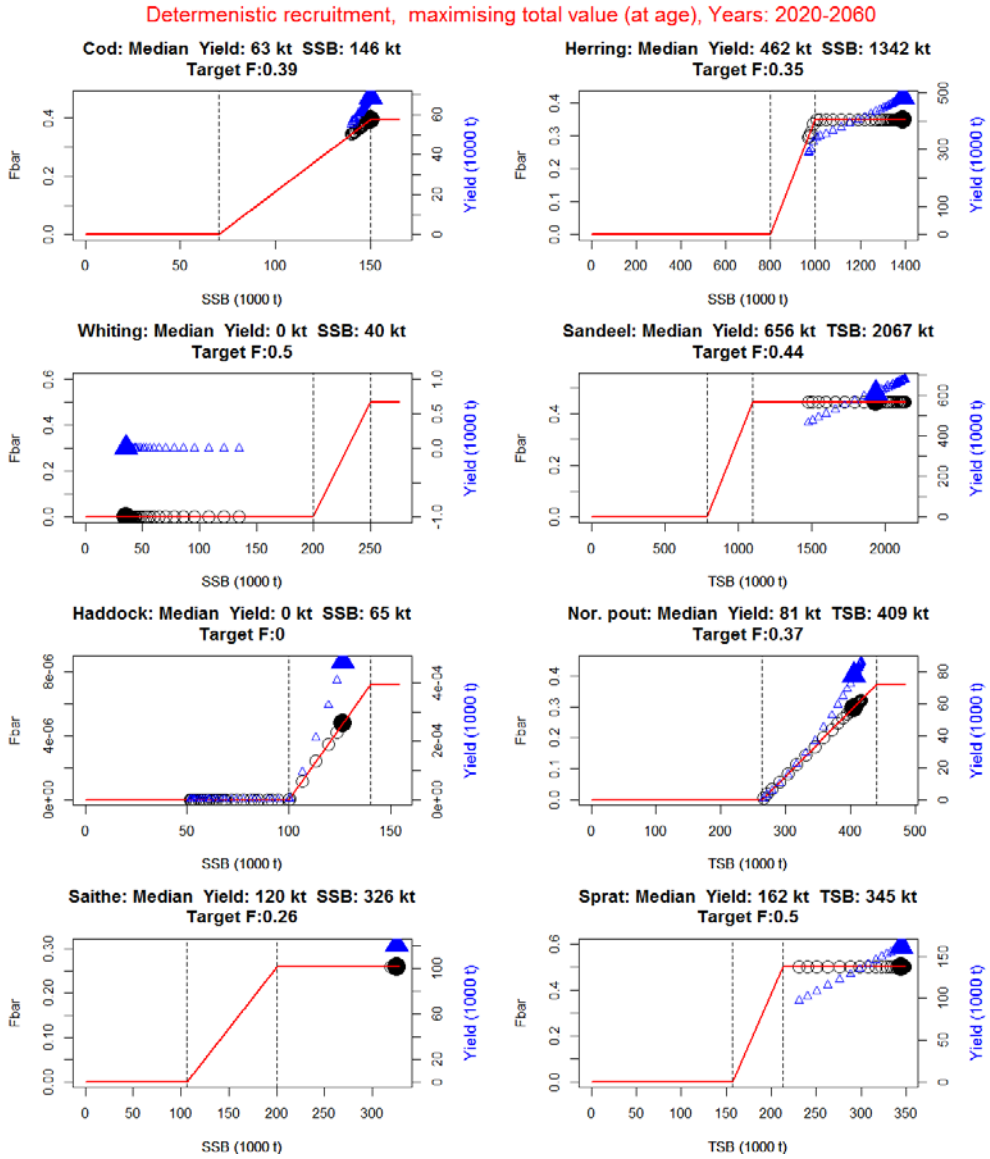



**Figure 10.5.7. Results of maximizing total value of the landings. The red line shows the HCR where the inflection points are given as input. The plateau of F (Fmsy) is estimated from maximizing the objective function (total value). The blue triangles show the yield by year, with the largest filled triangle as yield in the terminal year. SSB or TSB is shown in the same way using circles.**

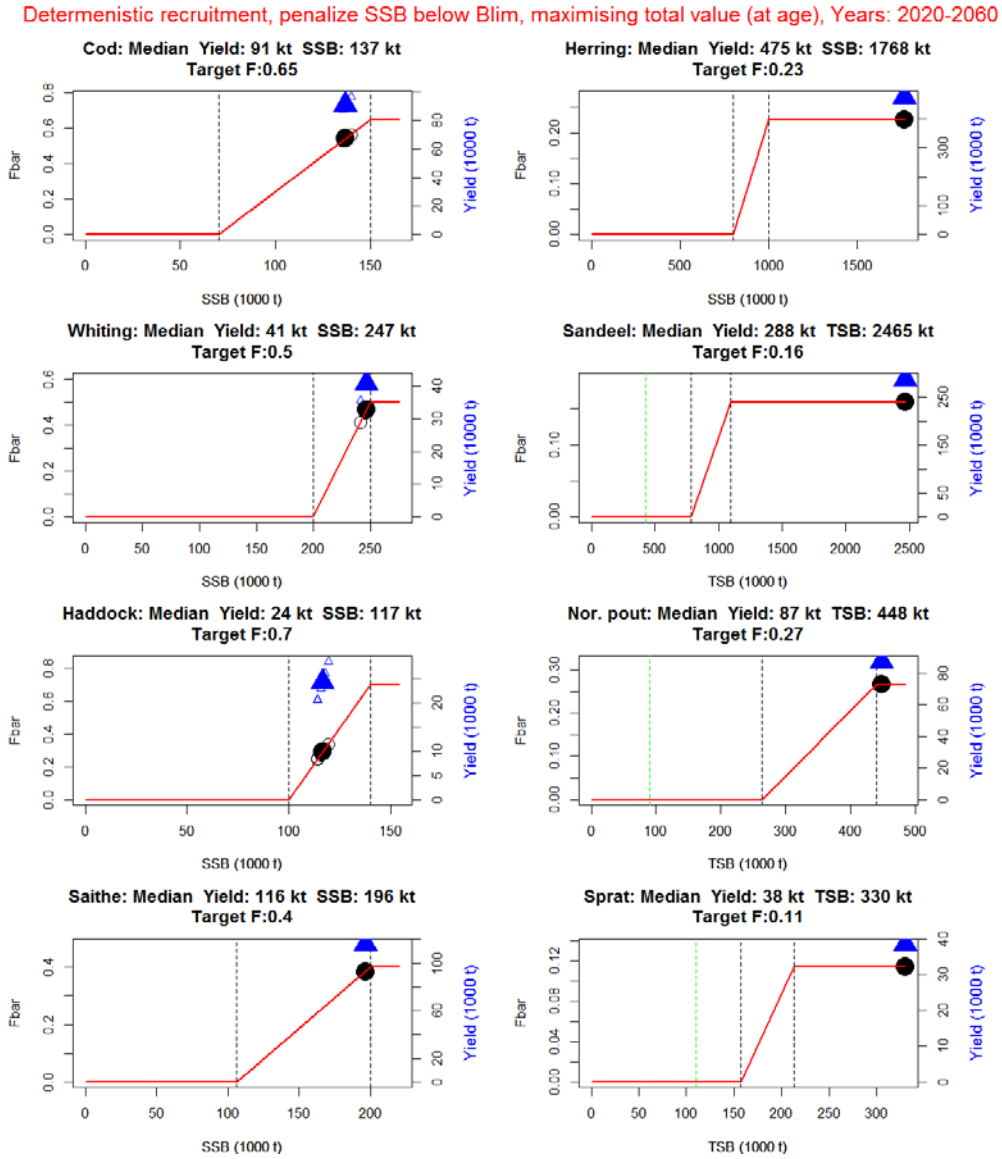



**Figure 10.5.8. Results of maximizing total value of the landings. The red line shows the HCR where the inflection points are given as input. The plateau of F (Fmsy) is estimated from maximizing the objective function (total value). The blue triangles show the yield by year, with the largest filled triangle as yield in the terminal year. SSB or TSB is shown in the same way using circles.**

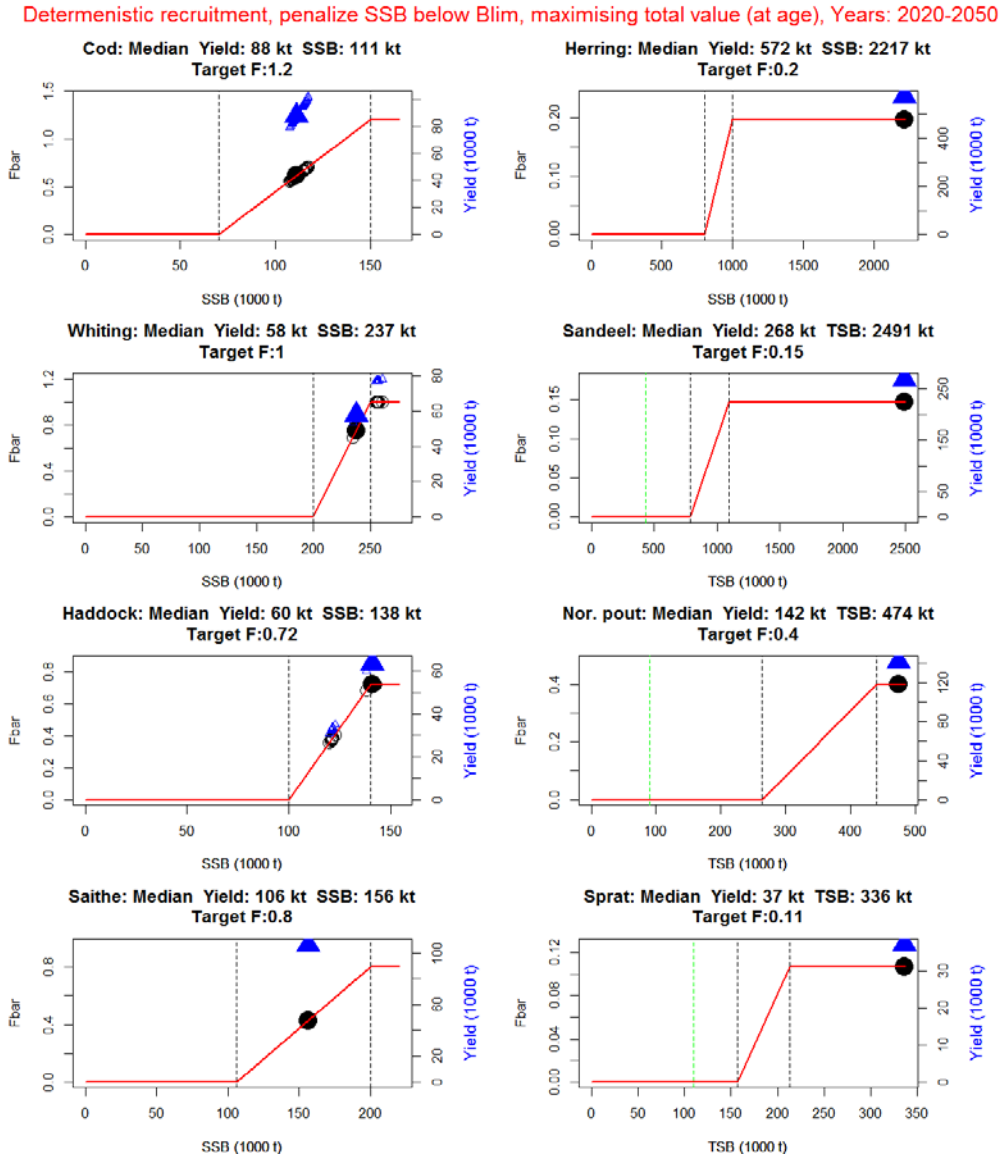



**Figure 10.5.9. Results of maximizing total value of the landings with unbounded Ftarget. The red line shows the HCR where the inflection points are given as input. The plateau of F (Fmsy) is estimated from maximizing the objective function (total value). The blue triangles show the yield by year, with the largest filled triangle as yield in the terminal year. SSB or TSB is shown in the same way using circles.**

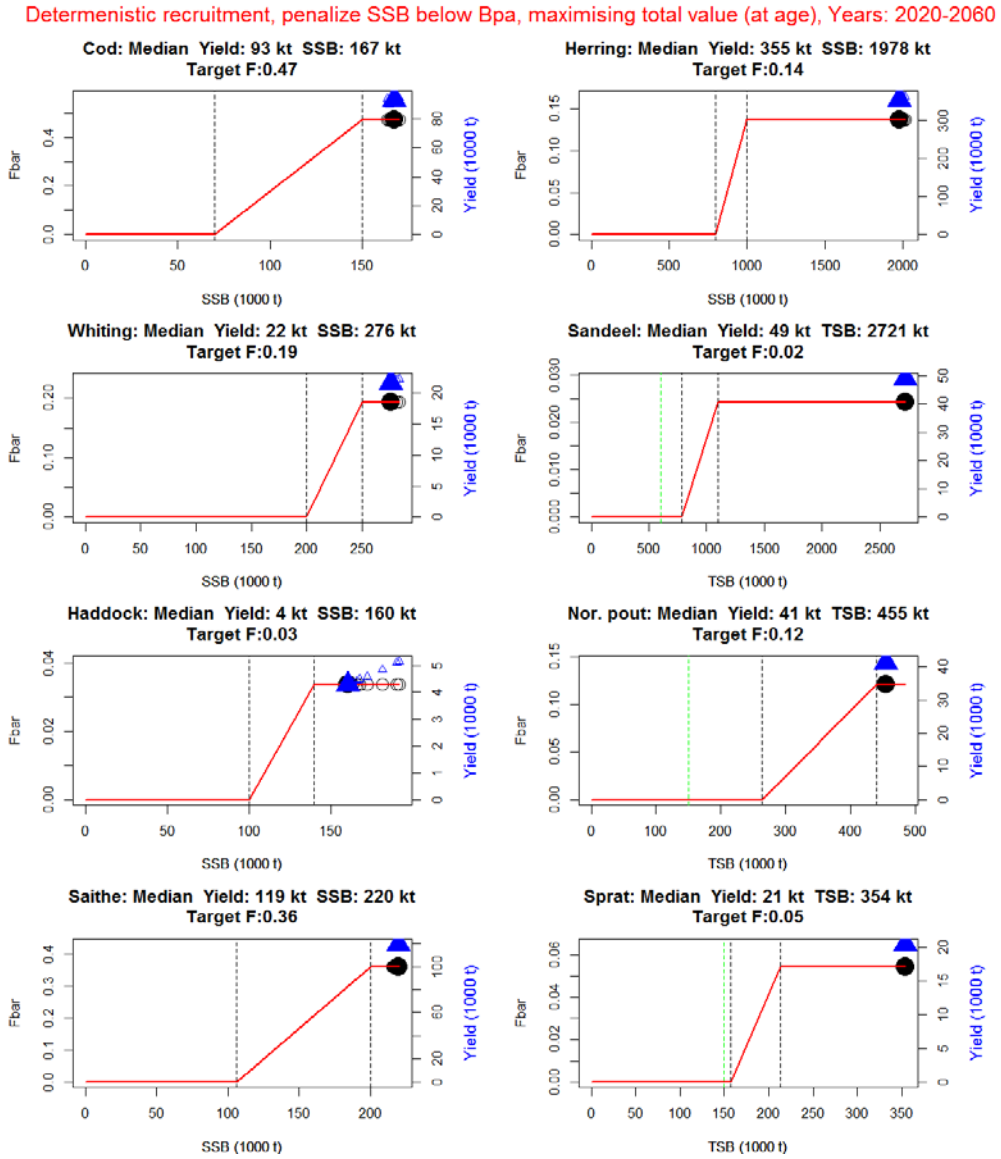



**Figure 10.5.10. Results of maximizing total value of the landings value and penalizing SSB below Bpa. The red line shows the HCR where the inflection points are given as input. The plateau of F (Fmsy) is estimated from maximizing the objective function (total value). The blue triangles show the yield by year, with the largest filled triangle as yield in the terminal year. SSB or TSB is shown in the same way using circles.**

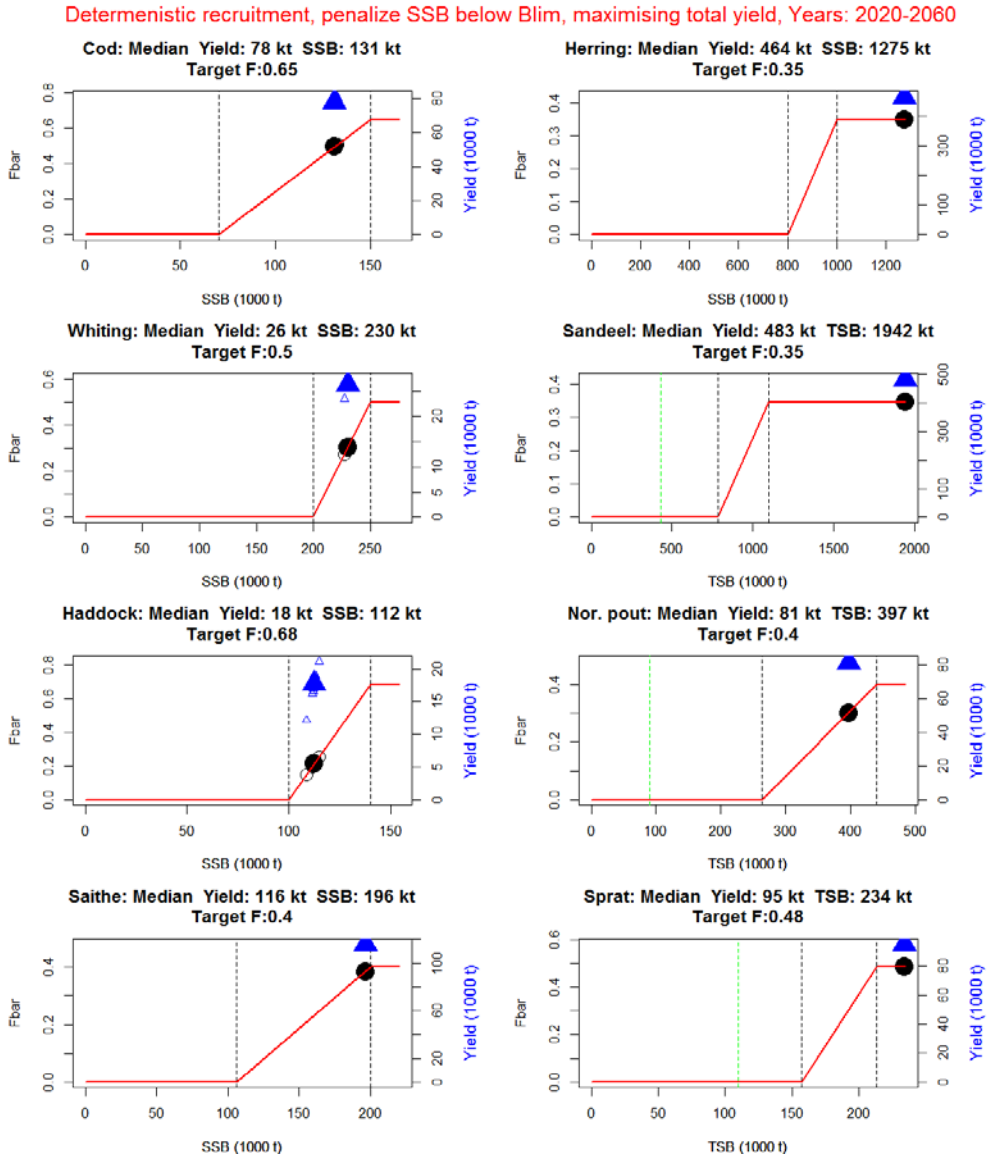



**Figure 10.5.11. Results of maximizing total yield. The red line shows the HCR where the inflection points are given as input. The plateau of F (Fmsy) is estimated from maximizing the objective function (total value). The blue triangles show the yield by year, with the largest filled triangle as yield in the terminal year. SSB or TSB is shown in the same way using circles.**

### 10.5.7 For which fishing mortalities is close-to-maximum average yield obtained for each of the modelled species when species interactions are included in a model?

The results from the previous section indicate that stock size of the predators, cod and saithe should be lower than the species-specific Bmsy to obtain an overall system MSY. Given a fixed F at 0.5 for cod and 0.45 for saithe, a rule based approach was tried to identify fishing mortalities close to maximum sustainable yield. The rule is simply a stepwise exclusion of the scenario F combination that provides the lowest yield relative to the estimated maximum yield for the particular species. The stepwise elimination of F combinations is outlined below:

a ) Exclude the F combination that provides the lowest species yield relative to the maximum yield calculated in any scenario.

b ) Recalculate median yield and identify the present maximum yield for all remaining F combinations.

c ) Repeat a), until all species have just one F combination (one F value) left.

The sequence of exclusions of F combinations is shown in Table 10.5.4. It is seen that the F value that gives the highest yield when all available F combinations are use is the same as the final F (FMSY). Using this method the FMSY values are estimated to

- Cod 0.50 (realized average F = 0.45)
- Whiting 0.60 (realized average F = 0.12)
- Haddock 0.40 (realized average F = 0.15)
- Saithe 0.45 (realized average F = 0.40)
- Herring 0.55 (realized average F = 0.40 )
- Sandeel 0.55 (realized average F = 0.48)
- Norway pout 0.60 (realized average F = 0.22)
- Sprat 0.55 (realized average F = 0.43)

It should be noted that the FMSY values are conditional of a SSB above Bpa (See Table 10.5.1). For SSB lower than Blim F is 0. There must be a linear reduction of F for SSB between Blim and Bpa. The realized average F values are considerably lower than target F for whiting, haddock and Norway pout. Figure 10.5.12 shows the relation between target F and realized F for the initial F-combinations used as basis for the stepwise exclusion of F. The discrepancy between target F and realized F is showing that SSB was considerably lower than Bpa for some species in most scenario years. The discrepancy between target and realized F may also indicate that the currently used SSB trigger values, or the type of HCR, are not appropriate.

The estimates of MSY and BMSY are presented in Table 10.5.6. These values represent the mean value for the period 2016-2070. SSB for cod, saithe, herring, sandeel and sprat are above Bpa, while SSB for haddock and Norway pout are between Blim and Bpa. The whiting SSB is below Blim. If whiting target F is fixed at 0.3, and the same exclusion rules are applied, the resulting Fmsy candidates (Table 10.5.7) are similar to the first set of FMSY, except for haddock, where Fmsy is now estimated to be 0.05 lower (0.35). SSB for whiting is increased by 2000 tonnes to 150000 t which is below Blim.

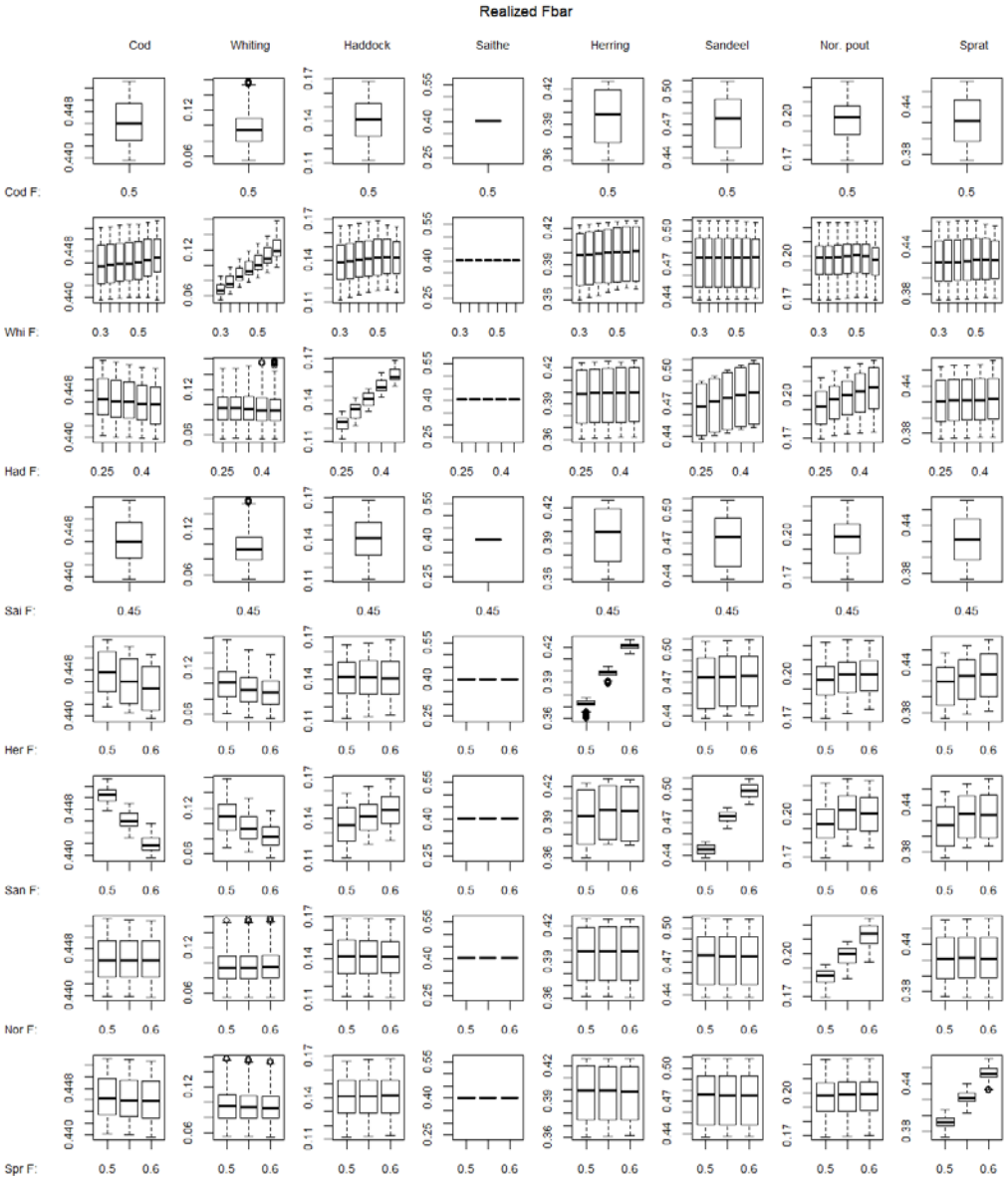



**Figure 10.5.12. Example of relation between target F (on the X-axis) for biomass above trigger point, and mean realized F (Y-axis).**

**Table 10.5.5. Sequence in reduction of F combinations that provide yield lower than the maximum yield.**

```
           0.2 0.25  0.3 0.35  0.4 0.45  0.5 0.55 0.6 0.65
Cod         NA   NA   NA   NA   NA   NA 1.00   NA  NA   NA
Whiting     NA   NA 0.65 0.72 0.78 0.84 0.90 0.95   1   NA
Haddock   0.91 0.94 0.96 0.98 1.00 1.00   NA   NA  NA   NA
Saithe      NA   NA   NA   NA   NA 1.00   NA   NA  NA   NA
Herring     NA   NA   NA 0.91 0.93 0.96 0.98 1.00  NA   NA
Sandeel     NA   NA   NA   NA 0.94 0.98 1.00 1.00   1 0.98
Nor. pout   NA 0.54 0.62 0.70 0.77 0.84 0.89 0.95   1   NA
Sprat       NA   NA   NA   NA   NA   NA 0.95 1.00  NA   NA
Nor. pout with F= 0.25 is below FMSY.

           0.2 0.25  0.3 0.35  0.4 0.45  0.5 0.55 0.6 0.65
Cod         NA   NA   NA   NA   NA   NA 1.00   NA  NA   NA
Whiting     NA   NA 0.64 0.72 0.78 0.84 0.89 0.94   1   NA
Haddock   0.91 0.93 0.96 0.98 1.00 1.00   NA   NA  NA   NA
Saithe      NA   NA   NA   NA   NA 1.00   NA   NA  NA   NA
Herring     NA   NA   NA 0.91 0.93 0.96 0.98 1.00  NA   NA
Sandeel     NA   NA   NA   NA 0.94 0.98 1.00 1.00   1 0.98
Nor. pout   NA   NA 0.62 0.70 0.77 0.84 0.89 0.95   1   NA
Sprat       NA   NA   NA   NA   NA   NA 0.95 1.00  NA   NA
Nor. pout with F= 0.3 is below FMSY.
Whiting with F= 0.3 is below FMSY. (leaving out MSY table)
Nor. pout with F= 0.35 is below FMSY.
Whiting with F= 0.35 is below FMSY.
Nor. pout with F= 0.4 is below FMSY.

           0.2 0.25  0.3 0.35  0.4 0.45  0.5 0.55 0.6 0.65
Cod         NA   NA   NA   NA   NA   NA 1.00   NA  NA   NA
Whiting     NA   NA   NA   NA 0.78 0.84 0.89 0.94   1   NA
Haddock   0.92 0.93 0.96 0.98 1.00 1.00   NA   NA  NA   NA
Saithe      NA   NA   NA   NA   NA 1.00   NA   NA  NA   NA
Herring     NA   NA   NA 0.91 0.94 0.96 0.98 1.00  NA   NA
Sandeel     NA   NA   NA   NA 0.94 0.98 1.00 1.00   1 0.98
Nor. pout   NA   NA   NA   NA   NA 0.84 0.89 0.95   1   NA
Sprat       NA   NA   NA   NA   NA   NA 0.95 1.00  NA   NA
Whiting with F= 0.4 is below FMSY.
Nor. pout with F= 0.45 is below FMSY.
Whiting with F= 0.45 is below FMSY.
Whiting with F= 0.5 is below FMSY.
Nor. pout with F= 0.5 is below FMSY.
Herring with F= 0.35 is below FMSY.
Whiting with F= 0.55 is below FMSY.
Haddock with F= 0.2 is below FMSY.

          0.25  0.3 0.35  0.4 0.45  0.5 0.55 0.6 0.65
Cod         NA   NA   NA   NA   NA 1.00   NA  NA   NA
Whiting     NA   NA   NA   NA   NA   NA   NA   1   NA
Haddock   0.95 0.96 0.98 0.99 1.00   NA   NA  NA   NA
Saithe      NA   NA   NA   NA 1.00   NA   NA  NA   NA
Herring     NA   NA   NA 0.94 0.96 0.98 1.00  NA   NA
Sandeel     NA   NA   NA 0.94 0.97 0.99 1.00   1 0.99
Nor. pout   NA   NA   NA   NA   NA   NA 0.95   1   NA
Sprat       NA   NA   NA   NA   NA 0.95 1.00  NA   NA
Sandeel with F= 0.4 is below FMSY.
Herring with F= 0.4 is below FMSY.
Sprat with F= 0.5 is below FMSY.
Nor. pout with F= 0.55 is below FMSY.
Herring with F= 0.45 is below FMSY.

          0.25  0.3 0.35 0.4 0.45  0.5 0.55 0.6 0.65
Cod         NA   NA   NA  NA   NA 1.00   NA  NA   NA
Whiting     NA   NA   NA  NA   NA   NA   NA   1   NA
Haddock   0.97 0.99    1   1 1.00   NA   NA  NA   NA
Saithe      NA   NA   NA  NA 1.00   NA   NA  NA   NA
Herring     NA   NA   NA  NA   NA 0.98    1  NA   NA
Sandeel     NA   NA   NA  NA 0.97 0.99    1   1 0.99
Nor. pout   NA   NA   NA  NA   NA   NA   NA   1   NA
Sprat       NA   NA   NA  NA   NA   NA    1  NA   NA
Haddock with F= 0.25 is below FMSY.
```

**Table 10.5.5 continued.**

```
           0.3 0.35 0.4 0.45  0.5 0.55 0.6 0.65
Cod         NA   NA  NA   NA 1.00   NA  NA   NA
Whiting     NA   NA  NA   NA   NA   NA   1   NA
Haddock   0.99    1   1 1.00   NA   NA  NA   NA
Saithe      NA   NA  NA 1.00   NA   NA  NA   NA
Herring     NA   NA  NA   NA 0.98    1  NA   NA
Sandeel     NA   NA  NA 0.97 0.99    1   1 0.99
Nor. pout   NA   NA  NA   NA   NA   NA   1   NA
Sprat       NA   NA  NA   NA   NA    1  NA   NA
Sandeel with F= 0.45 is below FMSY.
Herring with F= 0.5 is below FMSY.
Sandeel with F= 0.65 is below FMSY.

           0.3 0.35 0.4 0.45  0.5 0.55 0.6
Cod         NA   NA  NA   NA 1.00   NA  NA
Whiting     NA   NA  NA   NA   NA   NA   1
Haddock   0.99 0.99   1 0.99   NA   NA  NA
Saithe      NA   NA  NA 1.00   NA   NA  NA
Herring     NA   NA  NA   NA   NA    1  NA
Sandeel     NA   NA  NA   NA 0.99    1   1
Nor. pout   NA   NA  NA   NA   NA   NA   1
Sprat       NA   NA  NA   NA   NA    1  NA
Haddock with F= 0.3 is below FMSY.
Sandeel with F= 0.5 is below FMSY.
Haddock with F= 0.45 is below FMSY.
Haddock with F= 0.35 is below FMSY.
Sandeel with F= 0.6 is below FMSY.

          0.4 0.45 0.5 0.55 0.6
Cod        NA   NA   1   NA  NA
Whiting    NA   NA  NA   NA   1
Haddock     1   NA  NA   NA  NA
Saithe     NA    1  NA   NA  NA
Herring    NA   NA  NA    1  NA
Sandeel    NA   NA  NA    1  NA
Nor. pout  NA   NA  NA   NA   1
Sprat      NA   NA  NA    1  NA
```

**Table 10.5.6. Estimates of yield (landings) and SSB for Fmsy candidates, with fixed target F=0.5 for cod and fixed target F=0.45 for saithe.**

| | FMSY | Lower trigger biomass ('1000 tonnes) | Higher trigger biomass ('1000 tonnes) | MSY ('1000 tonnes) | BMSY ('1000 tonnes) |
|---|---|---|---|---|---|
| Cod | 0.50 | 150 | 70 | 91 | 168 |
| Whiting | 0.60 | 200 | 250 | 12 | 148 |
| Haddock | 0.40 | 100 | 140 | 22 | 120 |
| Saithe | 0.45 | 106 | 200 | 132 | 207 |
| Herring | 0.55 | 800 | 1000 | 561 | 1302 |
| Sandeel | 0.55 | 787 | 1098 | 622 | 867 |
| Norway pout | 0.60 | 263 | 440 | 84 | 132 |
| Sprat | 0.55 | 157 | 213 | 151 | 220 |

**Table 10.5.7. Estimates of yield and SSB for Fmsy with fixed target F=0.5 for cod, fixed target F=0.45 for saithe and fixed target F for whiting at 0.30.**

| | FMSY (F for biomass above trigger point) | Lower trigger biomass ('1000 tonnes) | Higher trigger biomass ('1000 tonnes) | Mean realized F | MSY ('1000 tonnes) | BMSY ('1000 tonnes) |
|---|---|---|---|---|---|---|
| Cod | 0.50 | 150 | 70 | 0.45 | 91 | 168 |
| Whiting | 0.30 | 200 | 250 | 0.07 | 8 | 150 |
| Haddock | 0.35 | 100 | 140 | 0.14 | 22 | 128 |
| Saithe | 0.45 | 106 | 200 | 0.40 | 132 | 207 |
| Herring | 0.55 | 800 | 1000 | 0.40 | 561 | 1303 |
| Sandeel* | 0.55 | 787 | 1098 | 0.48 | 616 | 859 |
| Norway pout* | 0.60 | 263 | 440 | 0.22 | 82 | 130 |
| Sprat* | 0.55 | 157 | 213 | 0.43 | 151 | 221 |

***Trigger biomass refers to total-stock biomass.**

# 11 ToR i): Based on various assumptions about policy choices, suggest a format for multispecies advice for single species groups and present examples of such advice for areas where the necessary information is available

## 11.1 Assumptions about policy choices

### 11.1.1 Precautionarity

Policy choices were investigated by WKM-TRADE prior to WGSAM (WKM-TRADE 2012). This group advised that ICES should delimit the space compatible with sustainable exploitation and good environmental status and make explicit the trade-offs within this space. Policy decisions and compromises can be carried out by managers and stakeholders informed by the ICES analysis. WKM-TRADE considered that precautionarity has precedence over obtaining MSY.

### 11.1.2 $F_{MSY}$ in a multispecies environment

In a dynamic environment with species and technical interactions, no fisheries can exploit all populations at $F_{MSY}$. Thus WKM-TRADE agreed that when offering trade-offs, ICES can provide scenarios below $F_{MSY}$ for the exploitation of some populations. This will allow a policy choice to be made within the limits defined and explained by ICES. $F_{MSY}$ could thus be defined as a range, although the upper bound should not be seen as optimum solution for fisheries exploitation rates. $F_{MSY}$ should be seen as the upper bound for the target F (WKM-TRADE 2012). It is the role of ICES to set the value for $F_{MSY}$ through a transparent process using appropriate methods.

### 11.1.3 MSY management in relation to Good Environmental Status

EU member states are currently finalizing the national suggestions for indicators of Good Environmental Status. In addition to this process, both WGSAM, WGFE and WGECO continue work towards defining suitable indicators of GES, in particular the aspects of foodwebs (descriptor 4, ToR e) in this report). As ICES Member Countries and working groups provide more of these objectives, they should ideally be built into the delimitation of space for policy choices, and further define the ICES onion of precautionary, MSY and ecosystem approaches. Until these indicators arrive, the working group chose to report on 3 aspects of foodwebs, 1 of which is suggested in the MSFD (see ToR e). These indicators include:

1 ) Total and natural mortality of all prey species
2 ) The proportion of the biomass of demersal fish individuals longer than 15 cm which is found in individuals larger than 40 cm (LFI)
3 ) The biomass of forage fish, the pelagic community and the demersal community.

### 11.1.4 Scenarios investigated

WKM-TRADE suggested the following scenarios to be investigated:

1 ) Precautionary scenarios
2 ) Maximize the proportion of single species MSY on average across species within precautionary scenarios

3 ) "Extreme" scenarios which maximize each species in turn within precautionary scenarios

4 ) North Sea: Consider the impact of the recovery of cod within precautionary scenarios

5 ) In addition to these, WGSAM decided to consider

6 ) Combinations providing 'close-to-MSY' for all species

7 ) Maximize the sum of weight of landings

8 ) Maximize the sum of value of landings

Reports on the results of these exercises are given in Section 10.5.

## 11.2 Suggested format for multispecies advice for decision-makers

Multispecies advice to decision-makers needs to identify the choices and trade-offs to be made, and give the context to these decisions in addition to the more specific advice needed for assessment groups (see below). As well as the presenting overall model results, a short executive summary highlighting should be presented the key interactions and trade-offs. The summary should be presented in a well explained and understandable form, allowing non-experts to understand and use the advice. The advice should be supported by a range of detailed results that can be consulted to go into more depth on given issues. The advice should present the range of $F_{MSY}$ for each species that is precautionary to all species as well as the range of precautionary $F_{MSY}$ combinations which provide yield close to MSY for all species. It should also clearly identify the most important trade-offs and choices to be made in managing the multispecies system. The results presented to the decision-makers need only include scenarios which are precautionary for all fish, whereas results which cause one or more species to crash need not be included in figure. However, such combinations should be clearly marked as unprecautionary in tables etc. It is also valuable to indicate which range of stock sizes and fishing mortalities have been observed for each stock, as model results in this range are likely to be more reliable than results extrapolating to conditions that have not been observed. Finally, any other issues which have been highlighted by the multispecies analysis should be summarized and presented to decision-makers. For instance, adaptive multispecies management may be highlighted if food shortages for some species are anticipated.

In conclusion, the group decided that multispecies advice for decision-makers should ideally include:

1 ) A description of the most important species interactions of modelled species

2 ) Advice on community and foodweb indicators (including natural mortality) which require information on the status of more than one species

3 ) Advice on the combination of target Fishing mortalities producing precautionary results in a multispecies environment

4 ) Advice on the combination of target Fishing mortalities producing close-to-MSY in a multispecies environment

5 ) Advice on ranges of F observed historically and hence defining the range in which confidence in the model is high.

6 ) Advice on important interactions and trade-offs

7 ) Advice on any other relevant multispecies issues

## 11.3 Suggested format for multispecies advice for single species assessment groups

The multispecies advice for single species assessments needs to be in a form that is relevant to both annual update assessments and the more in depth benchmark assessment meetings. This means that concise, focused advice is needed to feed into the annual assessments, but that more wide ranging advice can be requested in preparation for benchmark meetings, highlighting issues and scenarios that should be investigated further in the single species context.

For annual update assessments, the advice will include text and figures that give the multispecies context to the single species assessment, which can be copied and pasted into the assessment report. This includes an overview of the most important predator–prey interactions as well as the historical development in community and foodweb indicators.

For benchmark meetings there is a greater range of possibilities, which will depend on the availability of ecosystem models and particular concerns for the given ecosystem. For all areas, a range of precautionary $F_{MSY}$ combinations yielding close-to-MSY would be useful. The need for these arise because the multispecies analysis gives a range of $F_{MSY}$ values for each species that are to some extent precautionary to all the species rather than just to the species examined. These can then be examined at the benchmark meeting to identify their impacts on single species assessment, and appropriate values can be selected without exceeding the proposed limits of the range. In addition, specific issues may arise in analysing the multispecies context for a particular ecosystem which should be considered in more depth by the benchmark meetings. For example the effects of food abundance on predator growth and/or maturation may be important to single species assessment. In such cases the multispecies advice should include enough details of this for the benchmark meeting to use to select scenarios to examine more fully.

In conclusion, the group decided that multispecies advice for single species assessment or benchmark groups should ideally include:

1 ) A description of the most important species interactions of modelled species
2 ) Advice on community and foodweb indicators (including natural mortality) which require information on the status of more than one species. Tables of natural mortality and/or predator biomass should be available for download.
3 ) Advice on the combination of target Fishing mortalities producing precautionary results in a multispecies environment
4 ) Advice on the combination of target Fishing mortalities producing close-to-MSY in a multispecies environment
5 ) Advice on ranges of F observed historically and hence defining the range in which confidence in the model is relatively high.
6 ) Advice on important interactions and trade-offs
7 ) Advice on any other relevant multispecies issues

## 11.4 Guidelines for producing multispecies advice

### 11.4.1 Description of the most important species interactions of modelled species

The advice should include a description of the most important predators of each modelled species, their effect on the prey and vice versa. If available, this description should include descriptions of the contribution of each predator to prey mortality and the contribution of each prey to predator diet. If information exists linking predator performance to prey abundance, this should also be presented.

### 11.4.2 Advice on community and foodweb indicators (including natural mortality) which require information on the status of more than one species

The advice should ideally include information on the historical development of relevant community and foodweb indicators as well as the level of these indicators compatible with precautionary and close-to-MSY fishing mortality combinations. Where reference levels have been defined, the indicator value relative to the reference should be indicated in historic, precautionary and close-to-MSY evaluations.

### 11.4.3 Advice on the combination of target fishing mortalities producing precautionary results in a multispecies environment

Precautionary fishing mortality targets in a multispecies environment should preferably be defined using stochastic recruitment and a stepwise harvest control rule rather than deterministic recruitment and a flat $F_{MSY,}$ unless it can be demonstrated that there is no substantial difference between the results obtained by the two methods. The group considers this particularly important to avoid suggesting fishing mortalities which are unprecautionary at low biomasses resulting from poor recruitment success rather than overfishing. There were clear differences in the results of exploratory analyses comparing yield deterministic and stochastic simulations in the level of fishing mortality which could be considered precautionary, particularly for species with highly variable recruitment,. The stepwise harvest control rule should ensure a very low fishing mortality at biomasses below $B_{lim}$, a gradual increase in fishing mortality between $B_{lim}$ and the upper trigger point ($B_{MSYtrigger}$) and a constant target F above $B_{MSYtrigger}$. The resulting target fishing mortalities should only be considered precautionary when the full harvest control rule is used. Stochastic recruitment should be introduced using a stock–recruitment relationship which ensures that no recruitment occurs at a biomass of zero, with realistic variation around this relationship. The entire time-series should be used when estimating mean weight at age and stock recruitment relationship unless there is indication that parts of the time-series are less reliable or that major irreversible changes have occurred in the ecosystem.

Even when using a stepwise harvest control rule, some target F combinations can yield predictions where a substantial proportion of simulations result in biomasses of at least one species which is below $B_{lim}$. These combinations should also be classified as unprecautionary in a multispecies environment. Ideally, numerous stochastic simulations should be performed in order to accurately estimate the probability of falling below $B_{lim}$ for each combination of target fishing mortalities. Combinations for which this probability is greater than a specified risk level should be identified as unprecautionary.

The group considered it a particular challenge to present the result of all precautionary combinations of target fishing mortalities graphically in a manner that could be

understood by scientists and stakeholders without expert knowledge. WKM-Trade found the plots of yield of all species as a function of target fishing mortality for all species useful and understandable, and it was decided that this type of graphical representation should be included in the standard advice together with a thorough description on how to interpret the figure. An example of such a figure is given below, together with an explanation on how to interpret the figure.

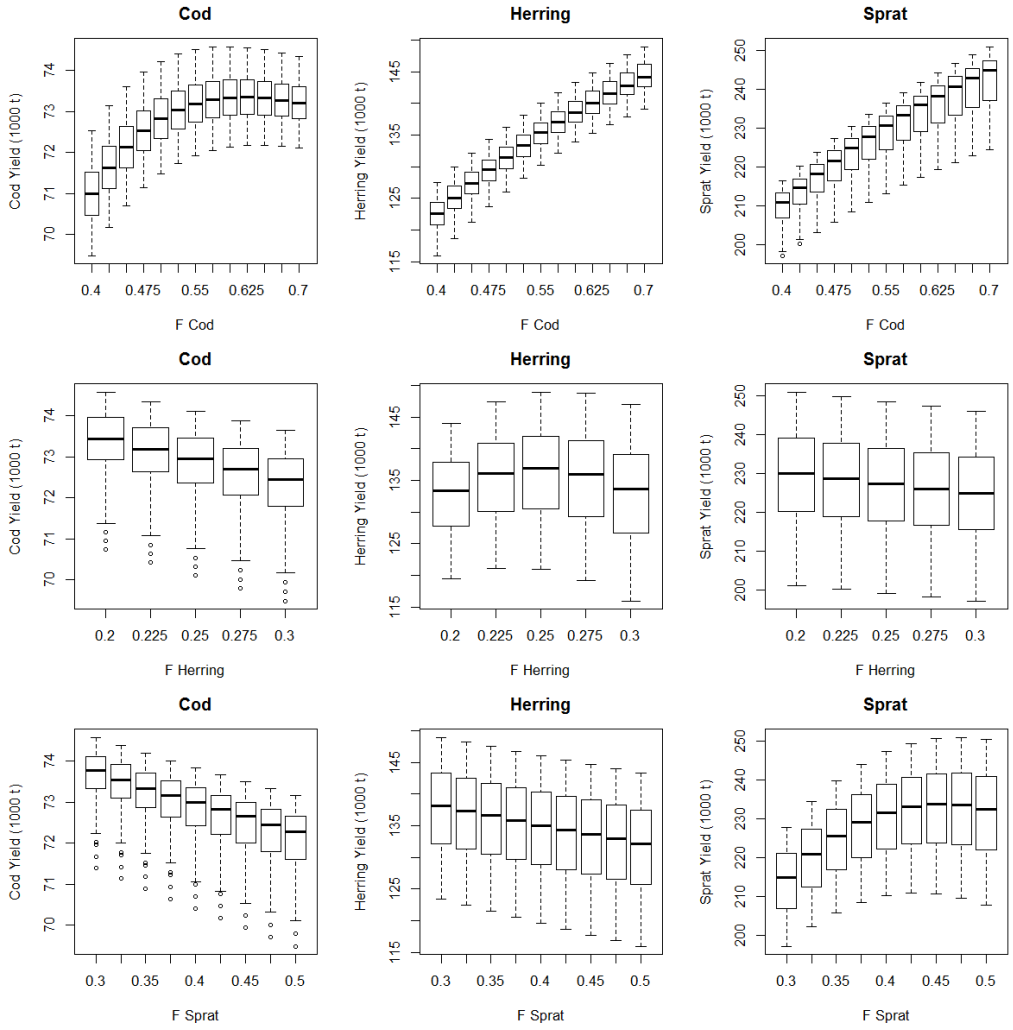



**Fig 11.4.1. Suggested format for graphical representation of yield in a multispecies environment in a system with three interacting species (one predator, cod, and two prey, herring and sprat). Each column represents the effect of varying target fishing mortality on all species in turn on yields of a specific species. Thus, the right column depicts the yield of sprat on the y-axis under different target fishing mortalities on cod (top), herring (middle) and sprat (bottom) on the x-axis.**

Solid black lines represent the median yield at the given target fishing mortality across all combinations of target F on other species. Boxes represent the range of yields derived when excluding the bottom 25% and the top 25% ranked yields. Whiskers represent the range of yields derived when excluding the bottom 5% and the top 5% ranked yields. Stars represent observations outside the range of 2sd. A wide box and whiskers implies that the yield of that species is heavily influenced by fishing pressure species other than the one depicted on the x-axis. In contrast a narrow range suggests that the yield is relatively insensitive to variations in fishing pressure on species other than that on the x-axis.

For example, the average yield at a target fishing mortality of 0.5 for cod across all precautionary combinations of target F for herring and sprat is 73 000 tons of cod, 132 000 tons of herring and 225 000 tons of sprat. The range of yields derived when the bottom 25% and top 25% range of yield (the boxes) and selecting a target fishing mortality for cod of 0.5 is 72 300 to 73 300 tons for cod, 129 000 to 134 000 tons for herring and 219 000 to 228 000 tons for sprat. The narrow range of variation for cod under cod fishing pressure (top left, box covers the median +/-0.7% at F=0.5) indicates that cod yield is relatively insensitive to fishing pressure on the other species. Conversely the wider variation for herring under changes in herring F (centre plot, box covers the median+/-4% at F=0.25) indicates that yield for this species is sensitive to fishing pressure on the other species.

The elements in the diagonal from the top left corner to the bottom right corner are similar to the way yield as a function of target F is represented in single species advice. Hence, they represent the change in yield of the specific species that we can obtain by changing the target fishing mortality on that species. Off-diagonal elements represent the effects of species interaction. For example, the change in mean yield of herring as cod target fishing mortality is increased shows the effect of the resulting lower cod stock, which then eats less sprat resulting in higher average yields of sprat. Further, the change in mean yield of cod as sprat target fishing mortality is increased shows the effect of the resulting lower sprat stock, which leads cod to eats less sprat and more cod resulting in lower average yields of cod. When the system contains more species, cases where increasing fishing on a top predator releases prey of this predator which then predate more heavily on their own prey resulting in lower stocks of the smallest prey can also occur (trophic cascades).

Importance of interactions are judged from the range of yields at any given target F and the change in average yield with target F.

For any given plot, if all values of target F produce large ranges of yields (boxes are large) this interaction is not the dominant interaction and is unlikely to present significant trade-offs. More specifically, if the mean values (black horizontal lines) for all values of F lie within the range for the boxes for all other values of F, then this indicates that there is no clear effect of changing that particular F for that species. For example, changes in herring fishing pressure do not affect sprat yield substantially (centre right), since the mean values for all values of F are contained with the range of all the boxes. This test acts as a filter to exclude cases from consideration where the effect of changing F on a given species is greatly outweighed by the effects of changing F on the other species. Once these are excluded, a second consideration the importance of the specific interaction. If the slope of the means is almost flat, then there is no significant interaction caused. In contrast, where the slope of the means is steep (and the interaction has not been excluded by the previous test), then the interaction may be considered “dominant”, and are important to consider for fisheries management. Thus dominant interactions to be considered are those where (a) the effects of changing the given F produce marked changes in the mean yields, and (b) where this stands out clearly against the effects of changing F on the other species. In the above example, these criteria lead to the identification of important interactions between cod yield and fishing mortality of all three species, between herring yield and cod fishing mortality and between sprat yield and cod and sprat fishing mortality.

### 11.4.4 Advice on the combination of target Fishing mortalities producing close-to-MSY in a multispecies environment

The advice on $F_{MSY}$ in a multispecies environment should provide information on the combinations of target $F_{MSY}$'s which result in yields above a specified fraction of the maximum sustainable yield of the given species. Maximum sustainable yield of the given species should be derived only from the range of target F's classified as precautionary.

### 11.4.5 Advice on ranges of F observed historically and hence defining the range in which confidence in the model is high.

The advice should indicate in the final output tables which F target values are inside the historically observed range for each species. Outside this range, yield at a given target F should considered to have higher uncertainty.

### 11.4.6 Advice on important trade-offs

Important interactions are defined from the range of yields at any given target F and the change in average yield with target F. If all values of target F produce large ranges of yields this interaction is not the dominant interaction and is unlikely to present significant trade-offs. If different values of target F produce different ranges of yields (average yield at a given target F is not included the 25 to 75% quartile boxes of all other precautionary yields), the interaction is defined as dominant. Dominant interactions where the average yield changes substantial with target fishing mortality the interaction are defined as important to consider in management. The most important interactions are defined by dominant ranking interactions according to the change in yield over the change in F-target and selecting the top ranked combinations of species F and yield of the same or other species (see fig 11.4.1).

## 11.5 Multispecies advice for the North Sea

### 11.5.1 Description of the most important species interactions of modelled species

The North Sea fisheries target a range of interacting species, most of which are also important prey to non-target species such as grey gurnard, seabirds and marine mammals. As a result of this, the multispecies assessment model (SMS) includes a total of 7 prey stocks (Cod, haddock, whiting, herring, Norway pout, sandeel and sprat), 10 predatory fish stocks (Cod, haddock, saithe, whiting, North Sea mackerel, western mackerel, North Sea horse mackerel, western horse mackerel, grey gurnard and starry ray), 2 species which do not interact with other species (plaice and sole), 8 seabird species (fulmar, gannet, great black backed gull, guillemot, herring gull, kittiwake, puffin and razorbill) and 2 marine mammals (grey seal and harbour porpoise; Figure 11.5.1).

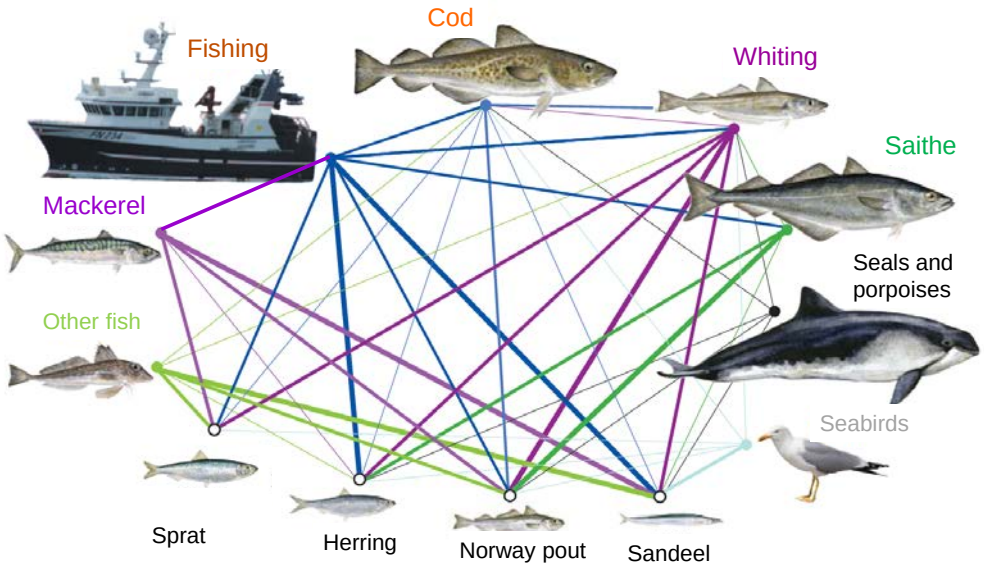



**Figure 11.5.1. Overview of the important predators on assessed North Sea fish species. Other fish include grey gurnard, North Sea and western Horse mackerel and starry ray. Seabirds include fulmar, gannet, great black backed gull, guillemot, herring gull, kittiwake, puffin and razorbill and seals and porpoises include grey seal and harbour porpoise. Colour of the lines indicates which predator the species is eaten by, the thickness of the lines indicate the biomass removed in this interaction (average from 1963–2010).**

Population dynamics of 10 stocks are modelled and change over time in estimations (cod, haddock, saithe, whiting, plaice, sole, herring, Norway pout, sandeel and sprat) whereas the other stocks are given as input (North Sea mackerel, western mackerel, North Sea horse mackerel, western horse mackerel, grey gurnard, starry ray, fulmar, gannet, great black backed gull, guillemot, herring gull, kittiwake, puffin, razorbill, grey seal and harbour porpoise). For the purpose of estimating yield at combinations of target fishing mortalities, all non-modelled predators are assumed to remain at the value observed in the last year.

The most important sources of mortality (sources which remove more than 10% of the biomass on the most important sources of mortality (sources which are responsible for more than 5% of the total mortality on average from 1963 to 2010) of each prey is given in Table 11.5.1. The most important food sources (sources constituting more than 5% of the diet on average from 1963 to 2010) of each predator are given in Table 11.5.2. The most important sources of mortality have varied over time as populations of predatory fish have shifted (Figure 11.5.3) as has the proportion of total mortality which is due to natural sources (Figure 11.5.4).

**Table 11.5.1. The most important food sources of each predator.**

| Predator | Food sources |
|---|---|
| Cod | Other (55%), haddock (13%), herring (8%), Norway pout (7%), whiting (7%) |
| Haddock | Other (80%), sandeel (15%), Norway pout (5%) |
| Saithe | Other (41%), Norway pout (19%), herring (17%), haddock (11%), whiting (7%), sandeel (5%) |
| Whiting | Other (60%), Norway pout (10%), sandeel (7%), sprat (6%), herring (6%), whiting (5%) |
| North Sea mackerel | Other (82%), sandeel (10%) |
| Western mackerel | Other (75%), sandeel (10%), sprat (7%), Norway pout (7%) |
| North Sea horse mackerel | Other (67%), sandeel (17%), sprat (13%) |
| Western horse mackerel | Other (92%), sandeel (7%), Norway pout (5%) |
| Grey gurnard | Other (61%), whiting (13%), sandeel (12%), Norway pout (9%) |
| Starry ray | Other (75%), sandeel (18%), Norway pout (5%) |
| Fulmar | Other (83%), sandeel (11%) |
| Gannet | Other (61%), sandeel (18%), herring (11%) |
| Great black backed gull | Other (89%), sandeel (5%) |
| Guillemot | Sandeel (42%), other (21%), sprat (14%), herring (14%), whiting (7%) |
| Herring gull | Other (92%) |
| Kittiwake | Other (63%), sandeel (28%), sprat (6%) |
| Puffin | Sandeel (55%), other (29%), sprat (8%), herring (8%) |
| Razorbill | Sandeel (37%), other (31%), sprat (22%), herring (9%) |
| Grey Seal | Sandeel (41%), other (38%), cod (9%), haddock (7%) |
| Harbour porpoise | Whiting (44%), other (36%), cod (14%) |

**These predators represent predators included in the SMS model of the North Sea. Additional predators are included in the North Sea EwE model and will be added in future.**

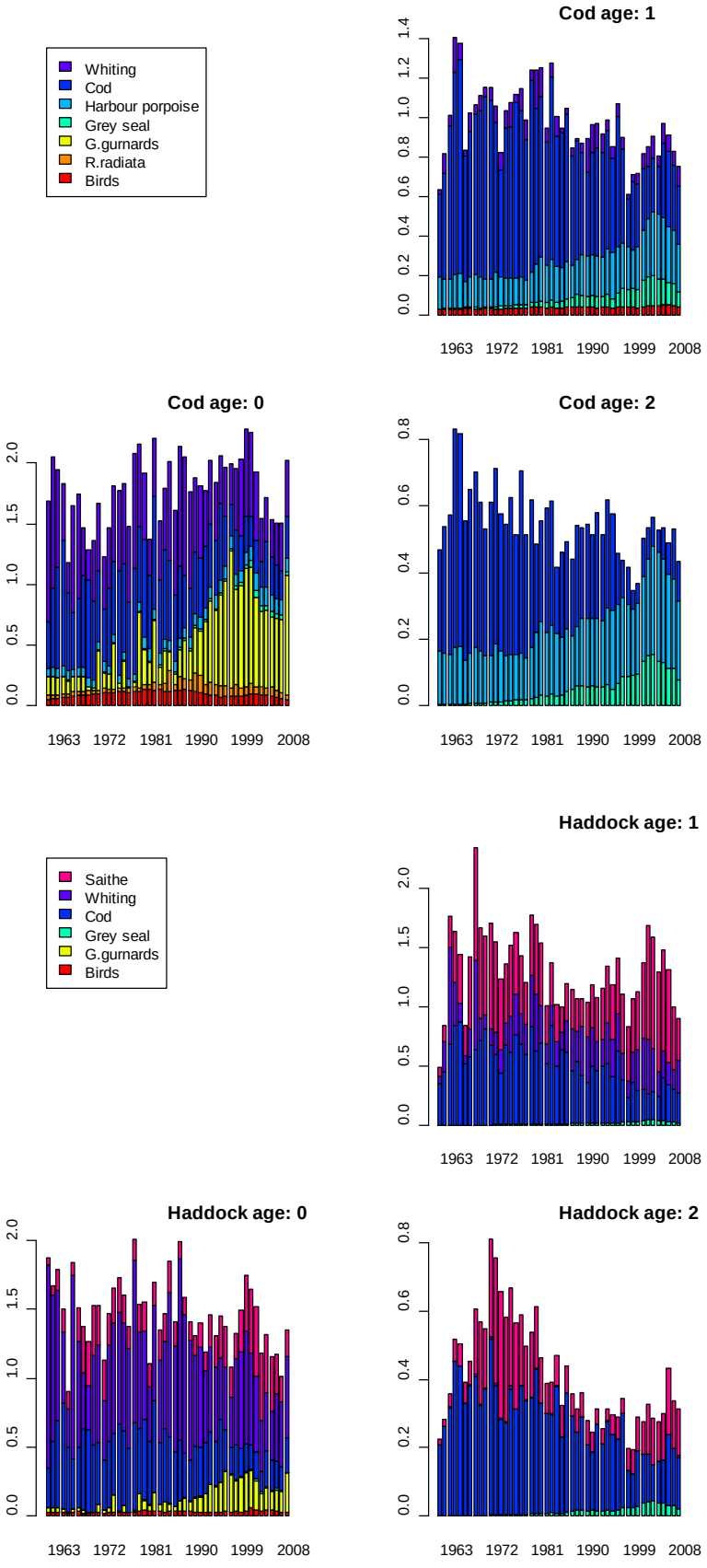



**Figure 11.5.2. Predation mortality (M2) by prey species and age inflicted by predator species. From WGSAM 2011.**

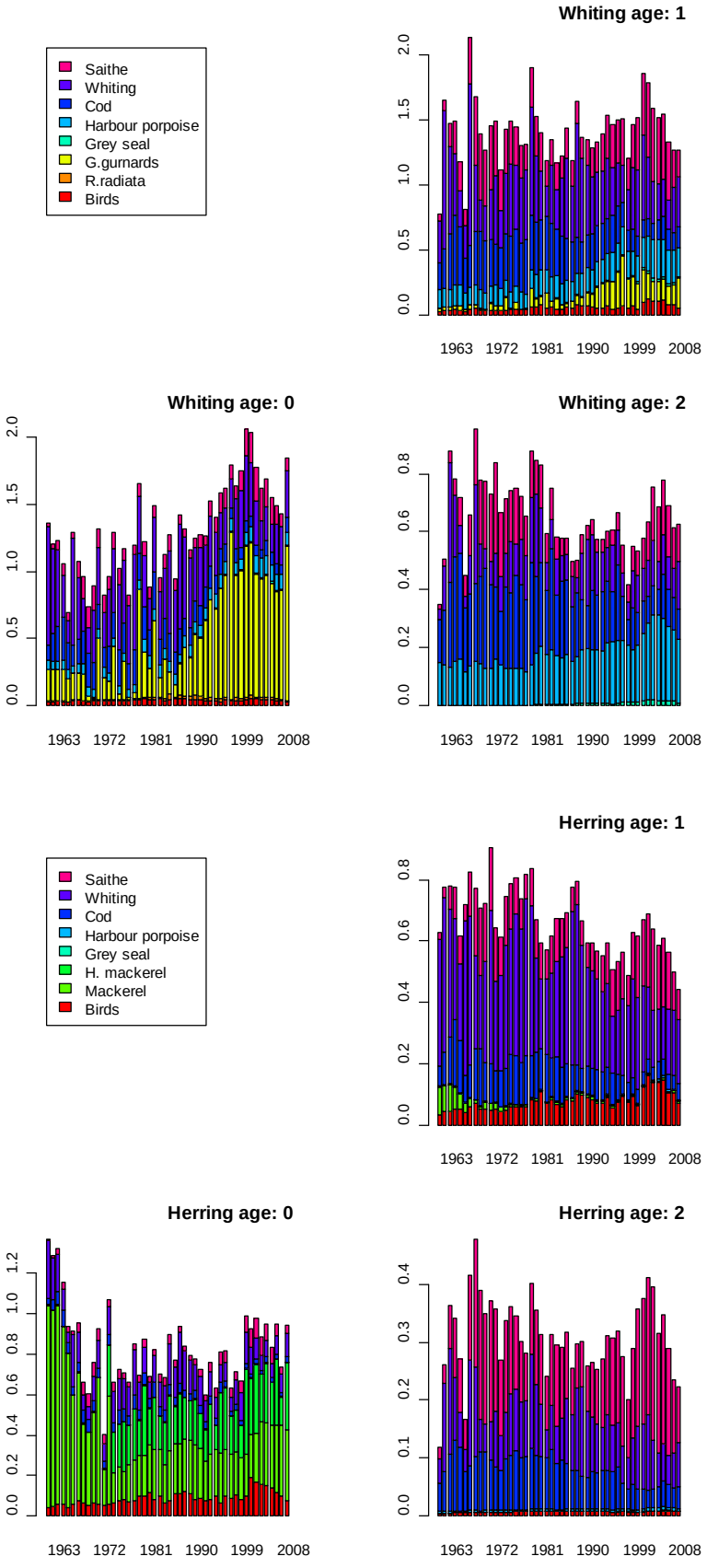



**Figure 11.5.2. cont. Predation mortality (M2) by prey species and age inflicted by predator species. From WGSAM 2011.**

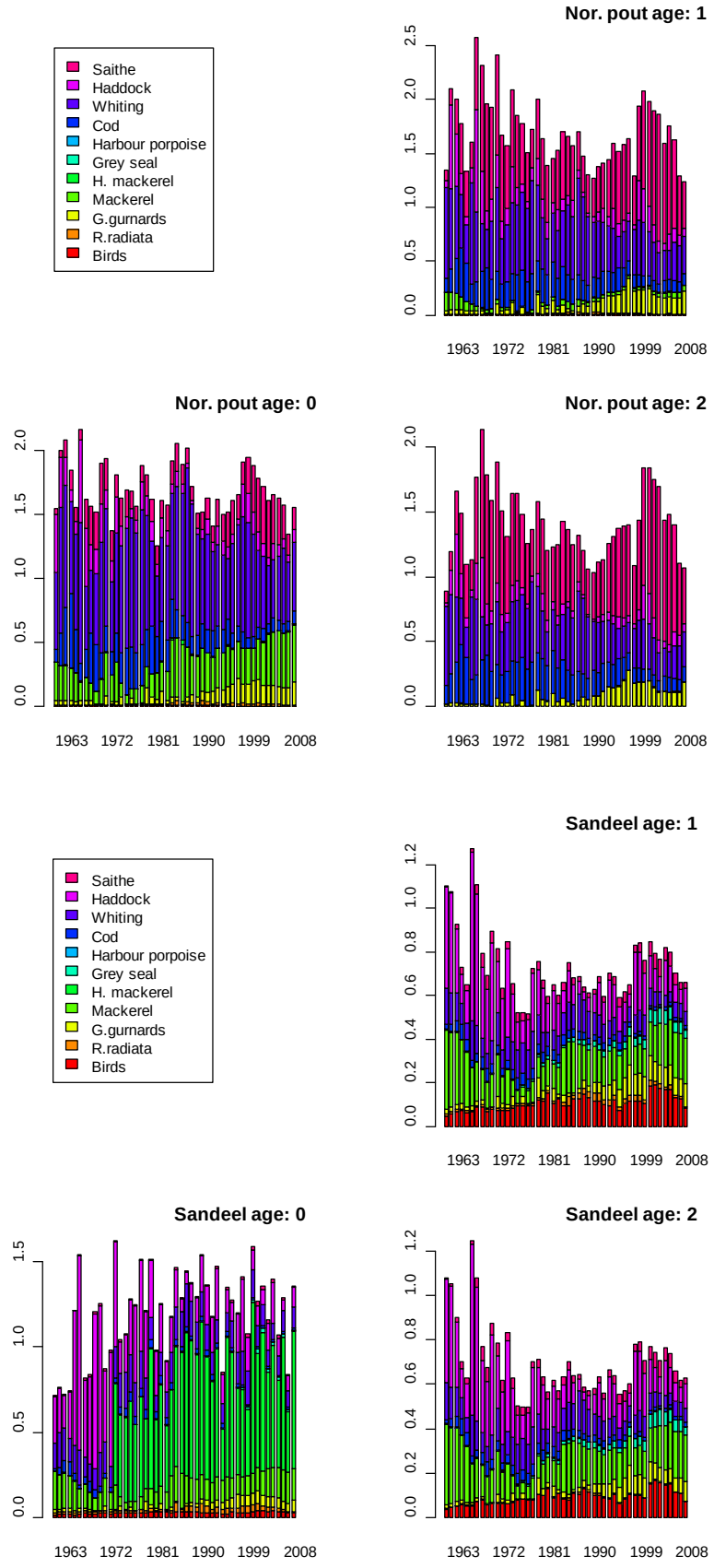



**Figure 11.5.2. cont. Predation mortality (M2) by prey species and age inflicted by predator species. From WGSAM 2011.**

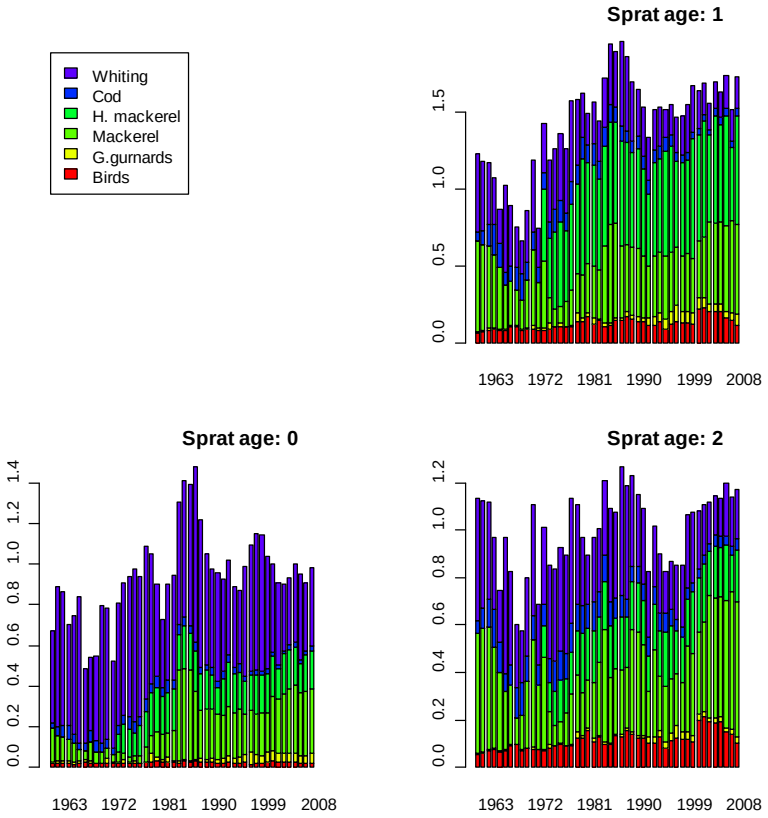



**Figure 11.5.2. cont. Predation mortality (M2) by prey species and age inflicted by predator species. From WGSAM 2011.**

#### 11.5.1.1 Effects of prey density on predator performance

Effects of prey density on predator performance are often difficult to detect, in particular for predators which are long lived, highly mobile or have a wide diet. Furthermore, many predators are likely to have evolved to cope with the high natural variation in particularly pelagic prey stocks. However, there are cases where such effects have been shown. Seabird breeding success often shows a close correlation with food fish abundance (Furness and Tasker, 2000; Rindorf *et al*., 2000; Davis *et al*., 2005; Frederiksen *et al*., 2005), whereas breeding numbers and adult survival may not track these short-term fluctuations (Boyd *et al*., 2006). Nevertheless, several recent studies do show a trade-off between adult survival rate and reproductive performance, as a result of adults increasing investment when food supply declines and so incurring costs (e.g. Davis *et al*., 2005). Most species of seabirds in the North Sea suffered delayed breeding and widespread reproductive failures in 2003, 2004, 2005 and 2006 (Frederiksen *et al*., 2004; Mavor *et al*., 2005, 2006, 2007; Reed *et al*., 2006). The most severe problems, including total failures of some species, occurred in Shetland and Orkney in the northernmost part of the North Sea. Although bad weather during the chick-rearing period was partly to blame at some colonies, the main proximate cause of the breeding failures was a lack of high-quality food (Davis *et al*., 2005; Wanless *et al*., 2005).

Fish species most likely to be affected by shortage of a particular prey species are the species for which this species make up a large proportion of the diet. Some of these

may be able to switch to alternative prey, and hence a large proportion in the diet is a necessary prerequisite but not necessarily sufficient to demonstrate a strong link.

### 11.5.2 Advice on community and foodweb indicators

Four community indicators were selected for presentation in the present version of multispecies advice: Natural mortality by age, percentage of total mortality caused by natural sources, the Large Fish Indicator and the biomass by guild. None of these suggested foodweb indicators have associated reference points yet as reference points for the Large Fish Indicator pertains only to survey catches and not assessment output. Hence, time-series of the different indicators are given which can be supplemented by the development in indicators relative to agreed reference points as these are defined.

#### Natural mortality by age

The development in natural mortality by age can be seen in Figure 11.5.3.

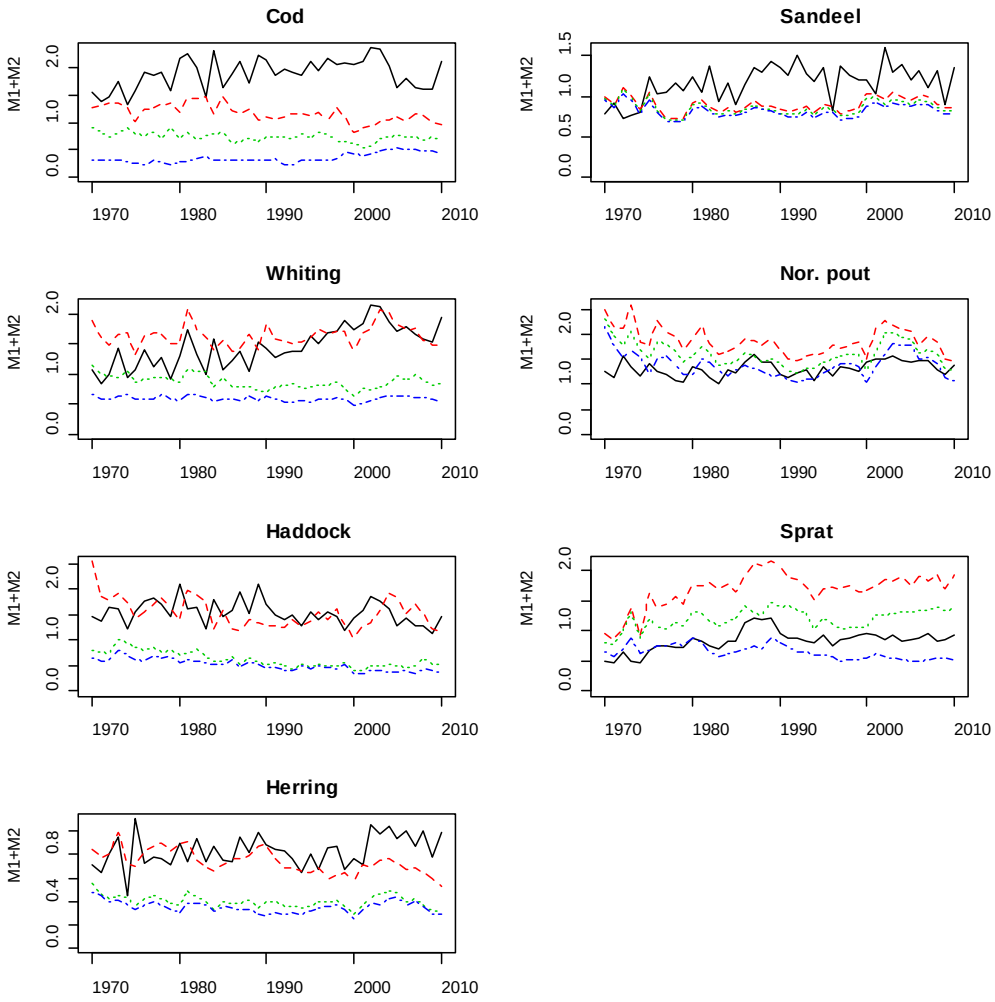



**Figure 11.5.3. Temporal development in natural mortality of age 0 (black) 1 (red), 2 (green) and 3 (blue).**

### Percentage of total mortality caused by natural sources

With the exception of cod and sprat, predation mortality is more than 50% of the total mortality in the majority of the years and predation mortality forms and increasing percentage of total mortality of all other species (Figure 11.5.4).

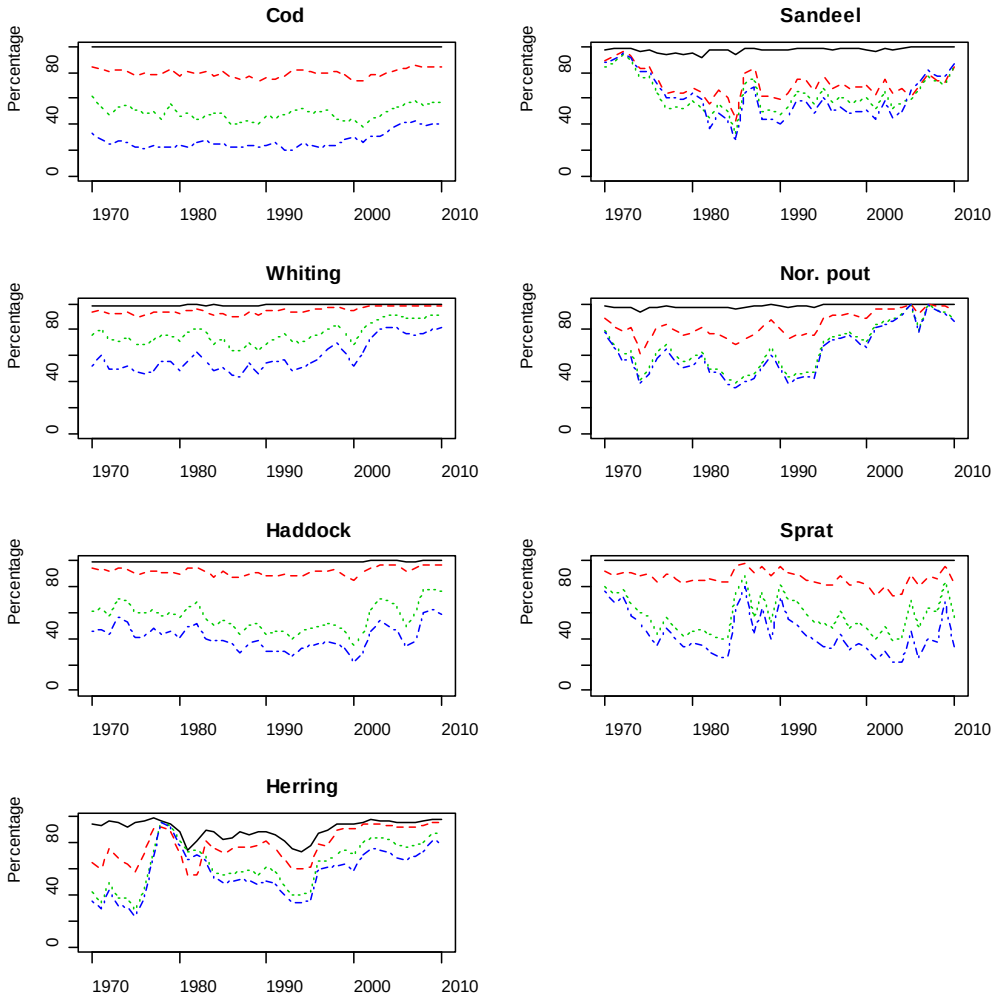



**Figure 11.5.4. Percentage of total mortality which is due to natural sources.**

## 11.5.3 Large Fish Indicator

The large fish indicator was estimated from assessment biomass at age assuming that all fish in a given age group have the average length of that age. This coarse assumption can be relaxed in future as length distributions of each age group are available for most species. Species included are cod, haddock, whiting, Norway pout, sole, plaice saithe, gurnard and starry ray.

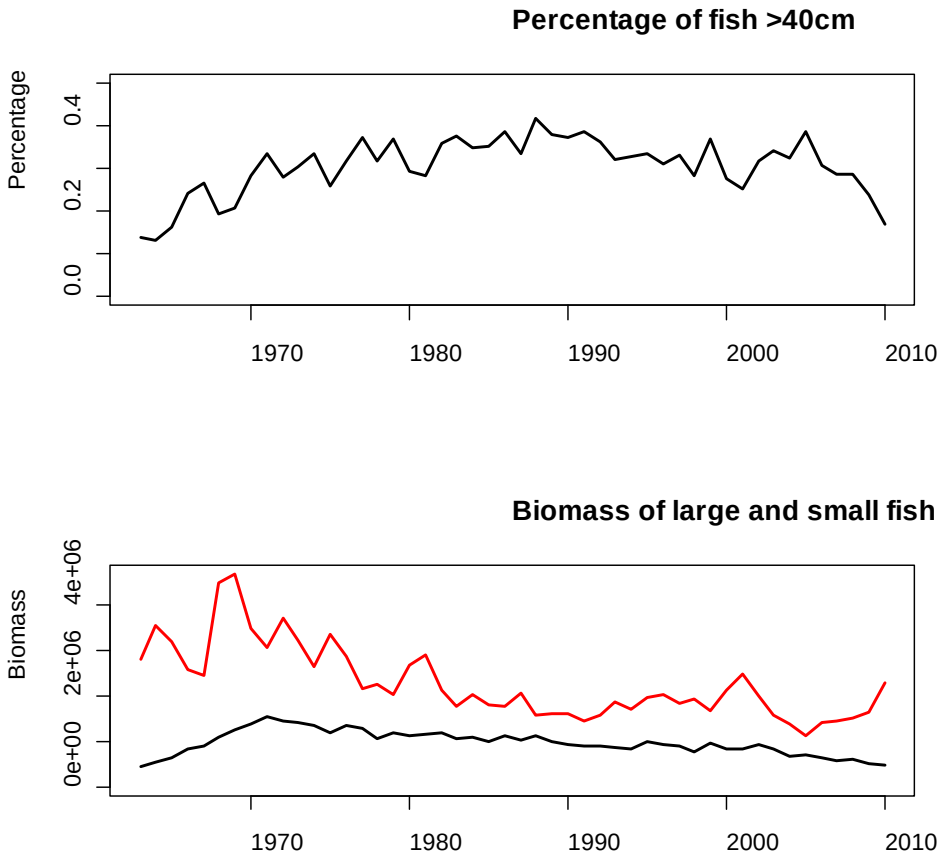



**Figure 11.5.5. Temporal development of the Large Fish Indicator estimated from assessment biomass at age. To demonstrate the causes of changes in the LFI, the biomass of small (red) and large (black) fish are given in the lower panel.**

### 11.5.4 Biomass by guild

Guilds were defined as forage fish (herring, sprat, Norway pout and sandeel), demersal fish (cod, saithe, haddock, whiting, grey gurnard, starry ray, plaice and sole) and migratory fish (mackerel and horse mackerel). The development in the biomass of each of these guilds are seen in Figure 11.5.6.

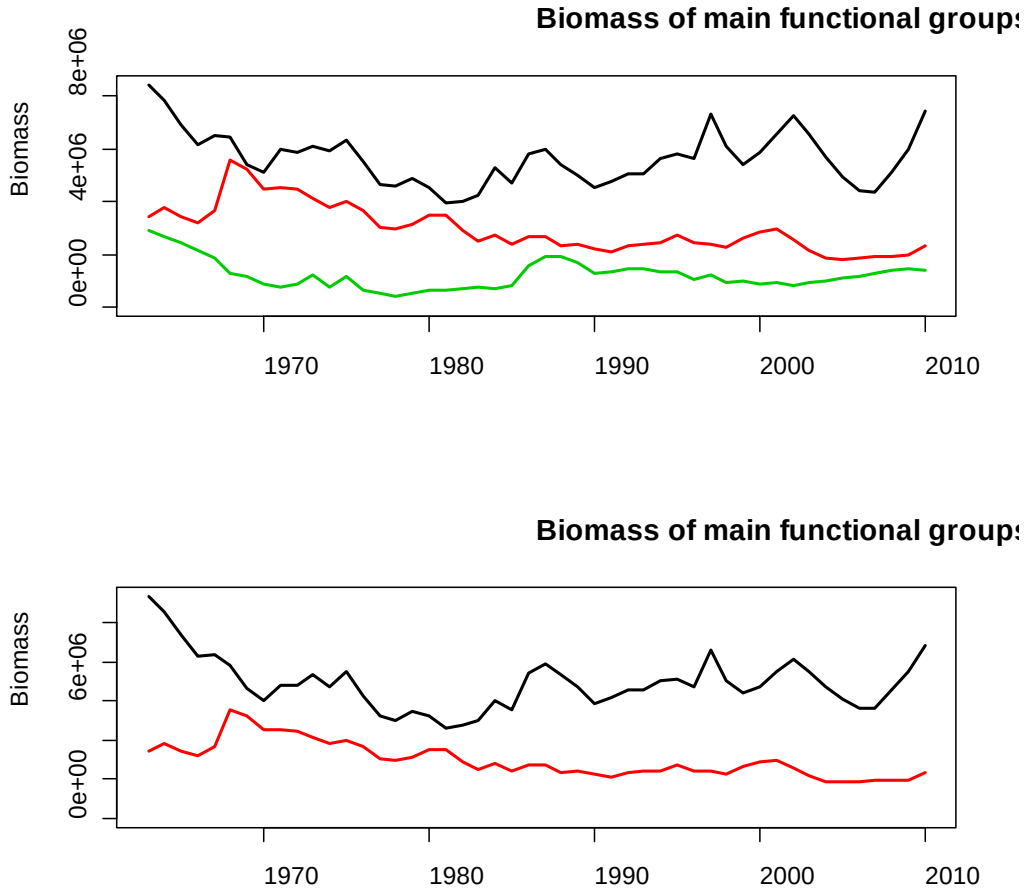



**Figure 11.5.6. Biomass of main functional groups (guilds). Upper figure: black = forage fish, red = demersal fish, green = pelagics (mackerel and horse mackerel). Lower figure: black = pelagics + forage fish, red = demersal fish.**

Tables of natural mortality and predator biomass are available for download at www.ices.dk/reports/SSGSUE/2011/WGSAM/SMS_summary.csv.

### 11.5.5 Advice on the combination of target fishing mortalities producing precautionary results in a multispecies environment

Scenarios of SSB and yield are shown in Figures 11.5.7 and 11.5.8. The SSB plot illustrates the direct and indirect responses to changes in F-values. As expected, the direct effect on an increasing F is a decreasing SSB (the diagonal of the figure). The indirect effect is however more complex. An increase in F on cod will give as smaller cod stock and less predation on its prey species. For whiting and haddock higher cod F leads to an increase in SSB (decrease in direct predation effect), while SSB of herring, sandeel, Norway pout and sprat will decrease (top row of Figure 11.5.8). This decrease in SSB for the typically prey species is due to that the gain in whiting and haddock SSB induces more predation on the prey species than the predation from a larger cod stock. The effects of an increase in saithe F are similar to the effects of an increased cod F.

There are also both direct and indirect predation effects of fishing on prey species. An increase in F on sandeel leads to a smaller sandeel biomass which induces a higher

cannibalism for cod and whiting leading to a decrease in SSB for those species. This lower biomass predates less on haddock which there therefore increases with increasing F on sandeel, although sandeel is the main (fish) prey for haddock. The effects of an increase in F on sandeel for the other prey species herring, Norway pout and sprat seems limited. Herring and Norway pout show a slight increase in SSB, probably because the decrease in cod and whiting biomass more than compensate the higher predation due to the reduction of sandeel.

The direct effect of F on yield (the diagonal of Figure 11.5.7) show a peak in yield for cod (F=0.45), haddock (F=0.3-0.4, range only precautionary if cod biomass is low) and sandeel (F=0.55) while the maximum yield for the other species is obtained at the outer range of F values presented. Whiting has the maximum yield at the highest F included (at 0.6) and saithe at the lowest F (at F=0.4). For herring, Norway pout and sprat the maximum yield is obtained at the highest F (at 0.55, 0.60 and 0.65 respectively). Given the selected range of F-values, the yield of all species except cod and saithe seems to be more affected by the indirect effect of fishing for other species (off diagonal figures) than by the direct effect of the individual species F (diagonal figures).

With target F at 0.4 on both saithe and cod, SSB of whiting is less than 80 kt (Figure 11.5.8) and thus far below any candidate for Blim. Maximum yield of cod is obtained for F in the range 0.45 to 0.50 (Figure 11.5.7) with median SSB above Bpa (Figure 11.5.8). Cod F at 0.5 seems therefor to be within precautionary limits for cod and to allow the whiting stock to increase. Saithe is another important predator on whiting. Saithe yield is highest (134 kt) for F=0.4, but F=0.45 gives practically the same yield (132 kt) with a median saithe SSB around 20kt less than for target F=0.4. Median saithe SSB for F=0.45 is slightly above Bpa (200 kt) such that F=0.45 seems precautionary. Figure 11.5.9 present SSB for cod F>=0.5 and saithe F>=0.45. Median SSB is above Blim (Table 11.5.2) for the given F level at the particular species. This median includes combination of F for other species, such that the minimum SSB for a particular F combination might be lower than Blim (Table 11.5.2 lower part) All the statistics are based on the median SSB over the years 2016–2070, which means that SSB in individual years might be below $B_{lim}$. More work is needed to provide improved performance statistics.

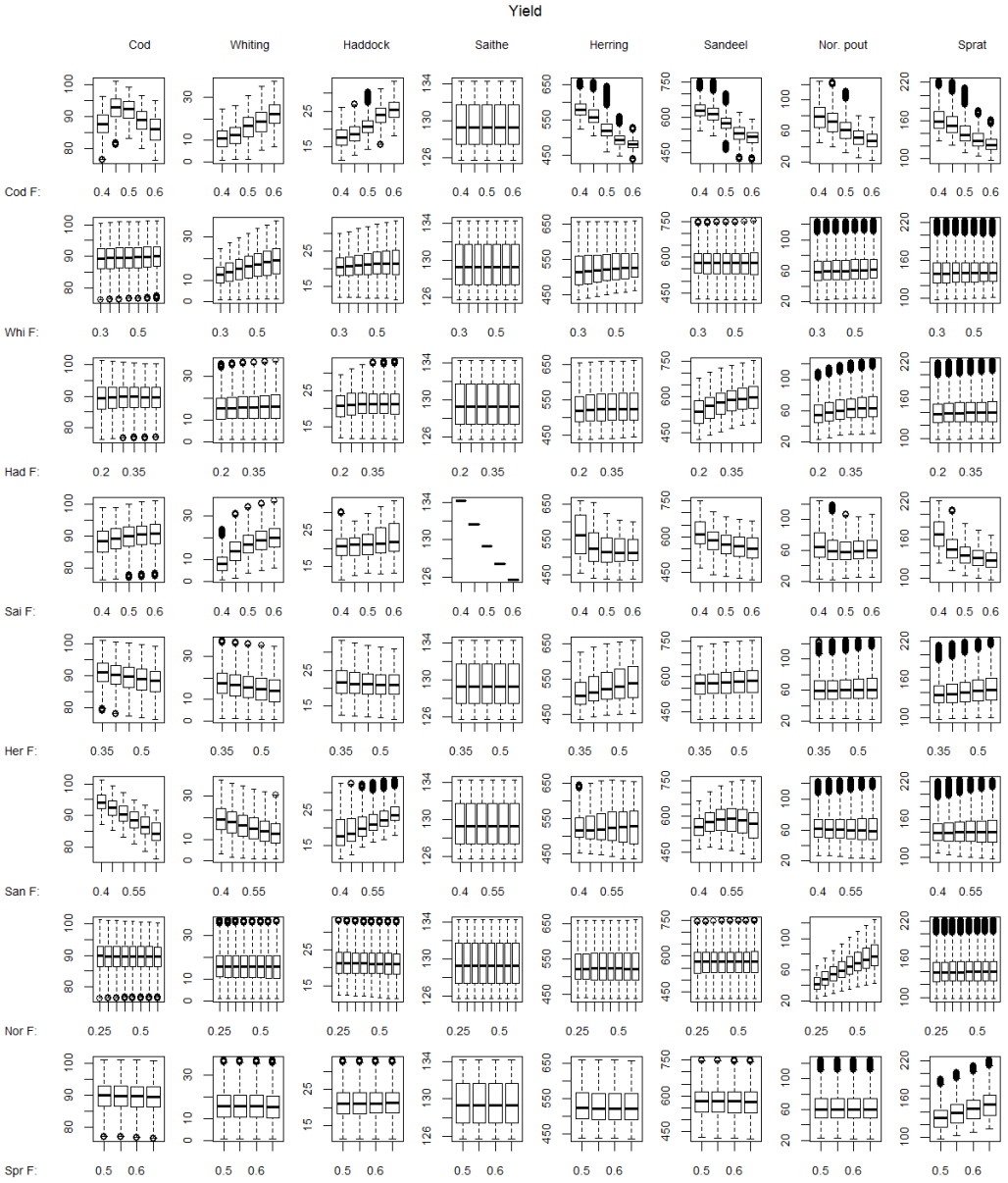



**Figure 11.5.7. Yield of (from left to right) cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat as a function of (top to bottom) target F on cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat. Note that the F-values on the X-axis is the target F for SSB above the specified MSY B trigger biomass. For SSB below the trigger value, the F realized might be considerably lower.**

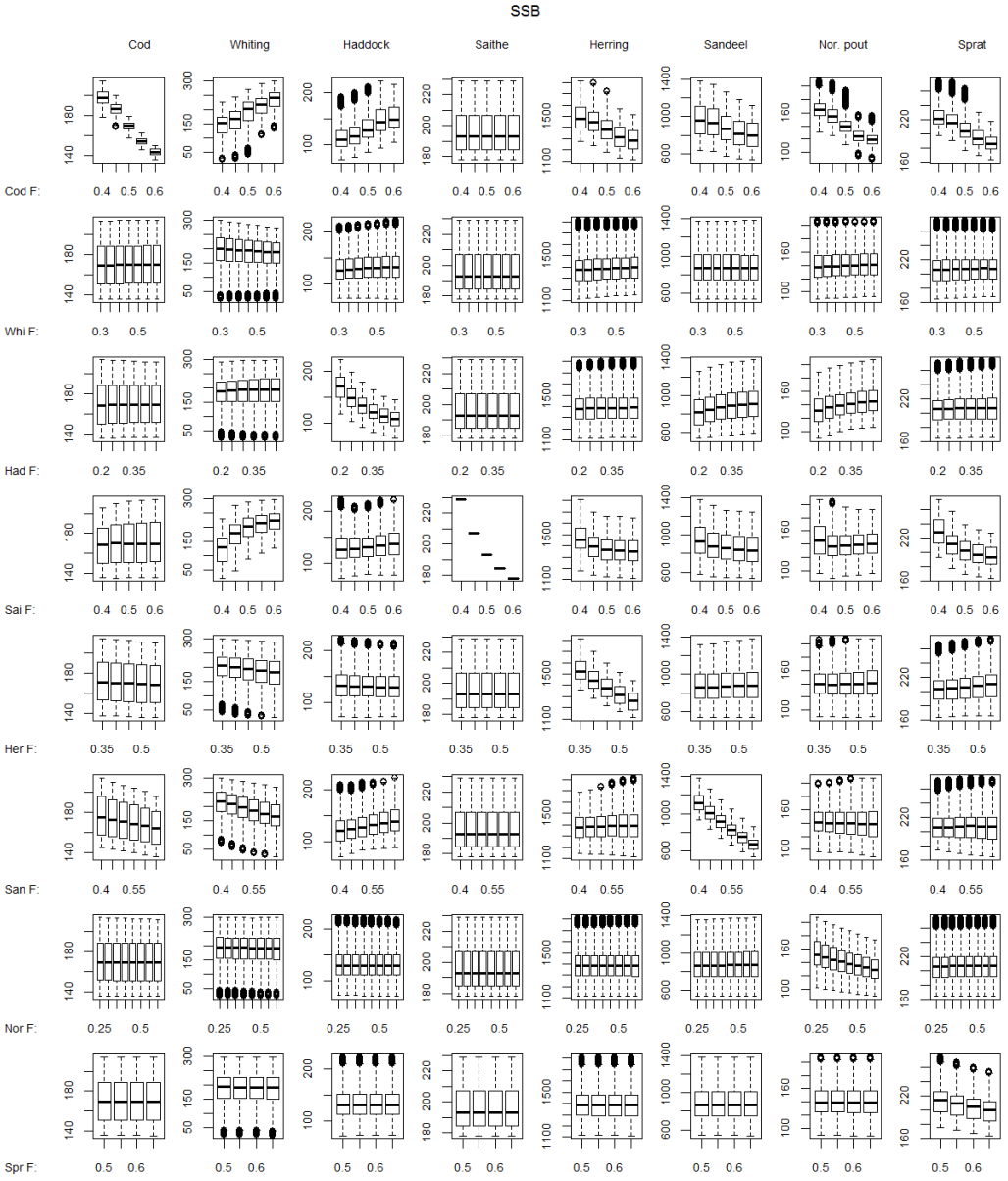



**Figure 11.5.8. SSB of (from left to right) cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat as a function of (top to bottom) target F on cod, whiting, haddock, saithe, herring, sandeel, Norway pout and sprat.**

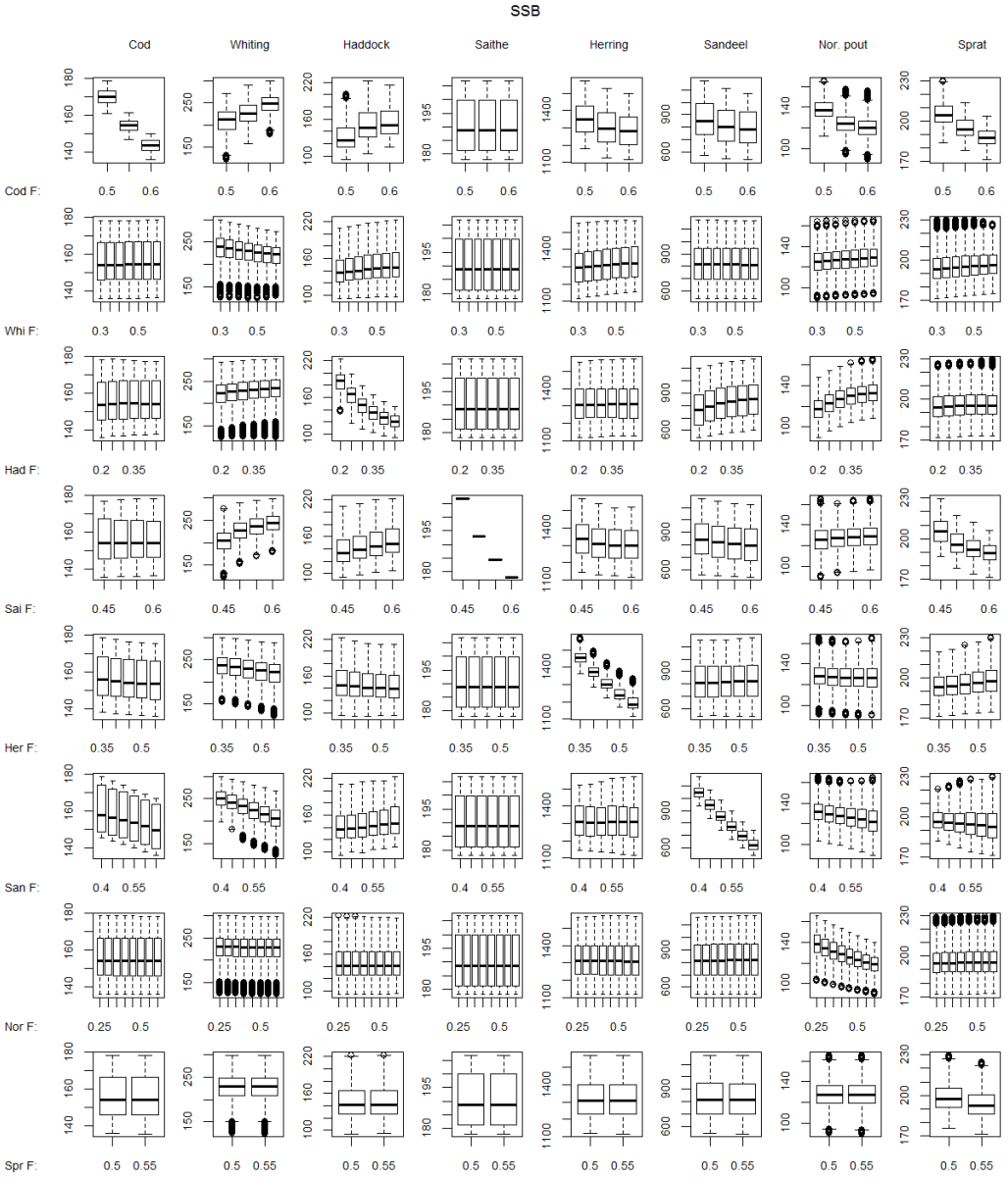



**Figure 11.5.9. SSB for cod F>=0.5 and saithe F>=0.45.**

### 11.5.6 Advice on the combination of target Fishing mortalities producing close-to-MSY in a multispecies environment

Stock size of the predators cod and saithe has a large influence on both yield and FMSY of other species. Hence, the choice of target fishing mortality for these species affect the yield of all other species. Estimates of MSY and BMSY are presented in Table 11.5.2. These values represent the mean value for the time period 2016–2070. SSB for cod, saithe, herring, sandeel and sprat are above Bpa, while SSB for haddock and Norway pout are between Blim and Bpa. Whiting SSB is below Blim. As FMSY values are conditional of a SSB above Bpa, FMSY is not realized for all stocks (Table 11.5.2). For SSB lower than Blim F is 0 and there is a linear reduction of F for SSB between Blim and Bpa. The realized average F values are considerably lower than target F for whiting, haddock and Norway pout. The discrepancy between target F and realized F is showing that SSB was considerably lower than Bpa for some species in

most scenario years. The discrepancy between target and realized F may also indicate that the currently used SSB trigger values, or the type of HCR, are not appropriate.

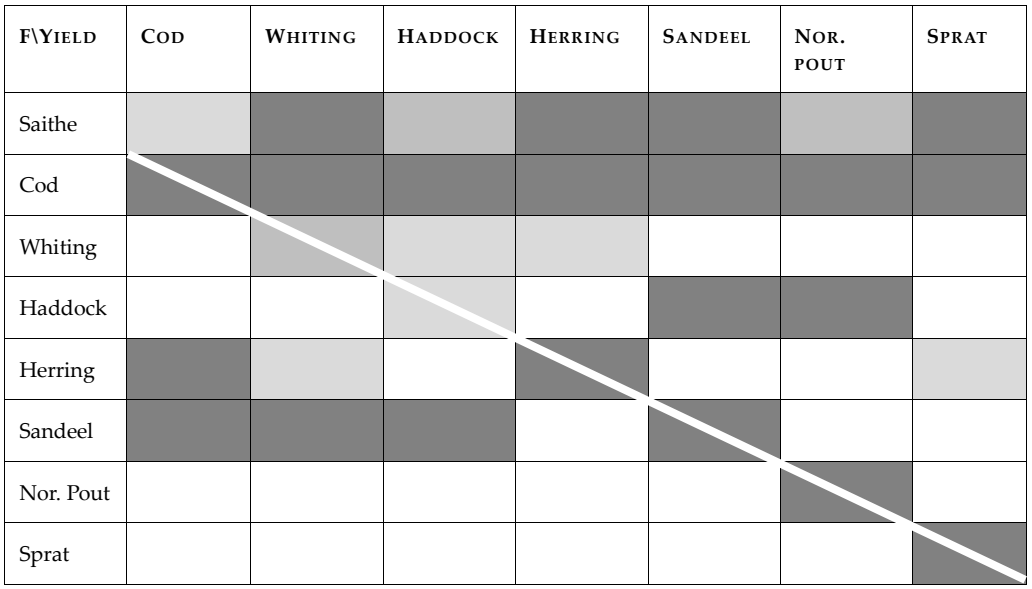
**Table 11.5.2. Estimates of yield (landings) and SSB for $F_{msy}$ candidates, with fixed target F=0.5 for cod, fixed target F=0.45 for saithe and fixed target F=0.30 for whiting. *: Trigger biomass based on TSB rather than SSB. Italics: Estimates of $F_{MSY}$ different from realized F as stock is frequently below $B_{PA}$.**

| | FMSY (F for biomass above trigger point) | Lower trigger biomass ('1000 tonnes) | Higher trigger biomass ('1000 tonnes) | Mean realized F | MSY ('1000 tonnes) | BMSY ('1000 tonnes) |
|---|---|---|---|---|---|---|
| Cod | 0.50 | 150 | 70 | 0.45 | 91 | 168 |
| Whiting | *0.30* | 200 | 250 | 0.07 | 8 | 150 |
| Haddock | *0.35* | 100 | 140 | 0.14 | 22 | 128 |
| Saithe | 0.45 | 106 | 200 | 0.40 | 132 | 207 |
| Herring | 0.55 | 800 | 1000 | 0.40 | 561 | 1303 |
| Sandeel* | 0.55 | 787 | 1098 | 0.48 | 616 | 859 |
| Norway pout* | *0.60* | 263 | 440 | 0.22 | 82 | 130 |
| Sprat* | 0.55 | 157 | 213 | 0.43 | 151 | 221 |

### 11.5.7 Advice on important interactions and trade-offs

Yield of virtually all species are strongly affected by the target F on cod and saithe and the effect of changes in predation on yield is much as large as or larger than the effect of target F on all species. A high F on cod and saithe releases the two smaller gadoids from predation and increases both biomass and yield of them. For haddock yield remains virtually unrelated to haddock F. However, for the remaining species, there is a reasonable relationship between species-specific F and yield. All species have at least two strong off-diagonal interactions, and whiting and sandeel yield are strongly affected by F on three other species (saithe, cod and sandeel and saithe, cod and haddock, respectively). Norway pout and sprat are the only two species where F does not significantly affect yield of other species. Whiting and haddock fishing mortalities are the second least important for other species with only two interactions each.

**Table 11.5.3. Importance of fishing mortality (rows) on yield (columns). Dark shading indicates high importance, light indicates low importance. Criteria: Dark: Median, upper and lower whiskers change, medium grey: median, upper and lower whiskers change but change is small, light grey: upper or lower whiskers change, white: not noticeable change in median or whiskers. White line indicates effects of species F on species yield.**




## 11.5.8 Advice on ranges of F observed historically and hence defining the range in which confidence in the model is high

The ranges of observed fishing mortalities from SMS are given in Table 11.5.4. Notes that these can differ from single species assessment estimates due to differences in the time period included and model formulation.

**Table 11.5.4. Historical range of observed fishing mortalities for each species.**

| Specioes | Lowest observedF | Highest observed F |
|---|---|---|
| Cod | 0.48 | 1.01 |
| Whiting | 0.15 | 0.87 |
| Haddock | 0.22 | 1.10 |
| Saithe | 0.18 | 0.73 |
| Herring | 0.01 | 0.93 |
| Sandeel | 0.05 | 1.23 |
| Norway pout | 0.00 | 1.59 |
| Sprat | 0.07 | 1.01 |

## 11.5.9 Advice on any other relevant multispecies issues

No advice is provided in this section.

## 11.6 Multispecies advice for the Baltic

### 11.6.1 Description of the most important species interactions of modelled species

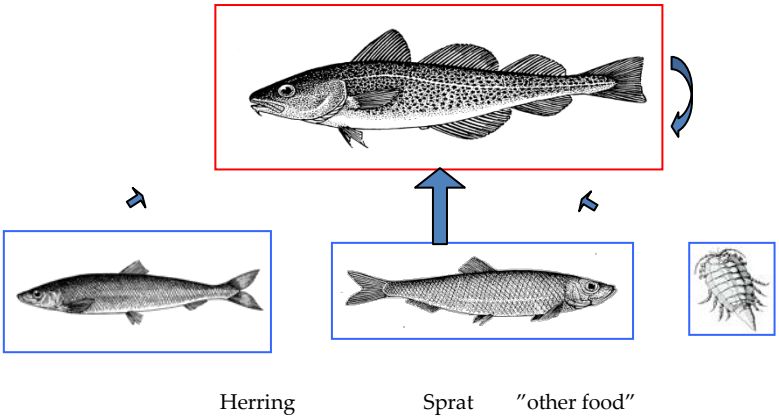



**Figure 11.6.1. Setup of the Baltic SMS model. Cod is the only predator, and forages on small cod, herring, sprat and zoobenthos which is pooled as 'other food'.**

Cod in the Baltic are cannibalistic. For the Baltic proper, cannibalism has been reported by Chrzan (1962), Stryzewska (1962), and Zalachowski (1977). The largest amounts of cod in cod stomachs have been observed in 1978 and 1979 (Zalachowski, 1986). Cod cannibalism is a relatively rare event. Of 3981 cod stomachs from the Bornholm Basin between 1963 and 1990, only 65 contained cod (Uzars and Plikshs, 2000). Nevertheless, in periods of high adult cod abundance, approximately 50% of a year class can be consumed by their con-specifics before reaching age 1 (Jensen and Sparholt, 1992, **I**). However, the suitability (Andersen and Ursin, 1977, Gislason and Sparre, 1987) of 0-group cod as prey for different cod age-groups differs substantially for cod age-groups 4+ when based on cod stomach content datasets from different periods of time (Figure 4).

Lishev and Uzars (1967) were the first to propose for the Eastern Baltic Sea that fluctuations in herring and sprat catches were caused by cod predation. Their findings, based on commercial catches and stomach data, were later substantiated by investigations on cod food composition in the Eastern Baltic (e.g. Uzars, 1994, Uzars and Plikshs, 2000; Bagge and Bay, 1987). More than 50% of the stomach content in Eastern Baltic cod greater than 40 cm is herring and sprat, and these two species contribute more than 80% of the total fish fraction in the diet of cod (Sparholt 1994). Only juvenile herring are preyed upon intensively by cod (Parmanne *et al.*, 1994).

**Table 11.6.1. The most important sources of mortality of each prey modelled.**

| Prey | Prey age | Important sources of mortality |
|---|---|---|
| Cod | 0 | Cod (67%) |
| Cod | 1 | Cod (42%) |
| Cod | 2 | Fishing (30%) |
| Cod | 3 | Fishing (66%) |
| Cod | 4+ | Fishing (79%) |
| Herring | 0 | Cod (35%) |
| Herring | 1 | Cod (28%), Fishing (17%) |
| Herring | 2 | Cod (20%), Fishing (34%) |
| Herring | 3 | Cod (15%), Fishing (44%) |
| Herring | 4+ | Fishing (56%) |
| Sprat | 0 | Cod (31%) |
| Sprat | 1 | Cod (44%), Fishing (11%) |
| Sprat | 2 | Cod (37%), Fishing (27%) |
| Sprat | 3 | Cod (34%), Fishing (34%) |
| Sprat | 4+ | Cod (31%), Fishing (38%) |

**Table 11.6.2. The most important food sources of each predator.**

| Predator | Food sources |
|---|---|
| Cod | Other (53)%, Sprat (32%), Herring (13%%),Cod (2%) |

#### 11.6.1.1 Documented effects of prey density on predator performance

Weight at age in eastern Baltic cod increased with decreasing stock size throughout the 1980s, and decreased throughout the 1990s despite continuously low stock size. This trend continued in the early 2000s. However, the slopes of the growth curves fitted to different cohorts did not differ significantly. Starting from about 30 cm total length, the cod diet changes from being almost exclusively based on benthic food to a mixture of benthos and fish, mainly clupeids.

For these reason, the analysis and modelling of cod growth consisted of two stages: first a model for age 3 weight at age dependent on clupeid availability, and second a model for growth in age-groups 3+.

Assumptions for the weight at age 3 models:

- Clupeid availability to age-group 2 cod during year *t-1* impacts upon weight of age-group 3 cod in the first quarter of year *t*
- Encounter rates between clupeids and cod increase with clupeid availability, and average cod growth is proportional to the average encounter rate.
- At 0 clupeid abundance, cod growth is not 0, but on a minimal (intercept) level, because the cod can partially compensate by foraging on benthic prey (Benthos abundance assumed constant).
- The relationship levels out at high clupeid abundance.

A simple Michaelis–Menten like model for weight at age 3 was chosen. The form was:

$$w = \frac{aN}{N+k} + b$$

Where $w$ is weight at age 3 in the first quarter of the year, N is clupeid biomass per predatory cod.

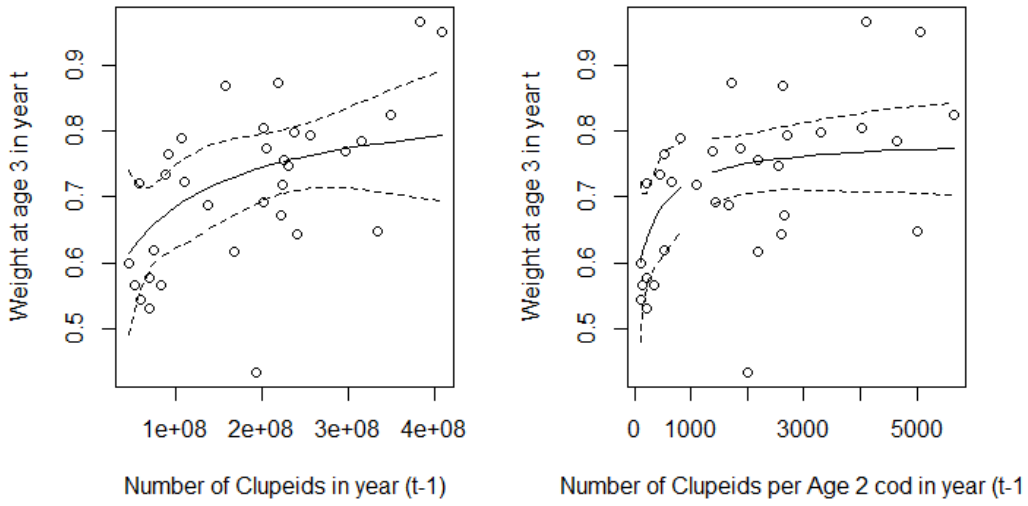



**Figure 11.6.2. Weight at age 3 for cod in year t vs. Absolute number of clupeids in year (t-1), left panel, and relative number of clupeids per Age 2 cod in year (t-1), right panel. The solid line is the model fit, stipled line 95% confidence limits of the model.**

The model is not significant. However, it is not to expect that the usage of clupeid abundance alone gives a strong signal. Alternatively, cod weight at age 3 was modelled dependent on number of clupeids divided by number of age 2 cod in the preceding year. Accounted for the number of age 2 cod yielded a significant intercept term $b$ ($P<0.01$). This might be an indication that there actually is a grazing effect of cod on clupeids, especially sprat. In the current stock situation, cod is concentrated in the Bornholm Basin, and here sprat acoustic surveys in May and October/November indicate a large decrease of the sprat biomass in the basin during the second half of the year. Whether or not this decrease really is a primary consequence of cod predation remains to be investigated further.

The model fit might be improved by using subarea specific clupeid and cod abundances, instead of aggregate abundance over the whole Eastern Baltic Sea.

The growth of age groups 3 to 7 is well described by von Bertalanffy growth parameters as can be seen in cohort-based Ford-Walford plots. Deviations from linearity in some years have to be investigated for the effect of differing prey availability to the cod.

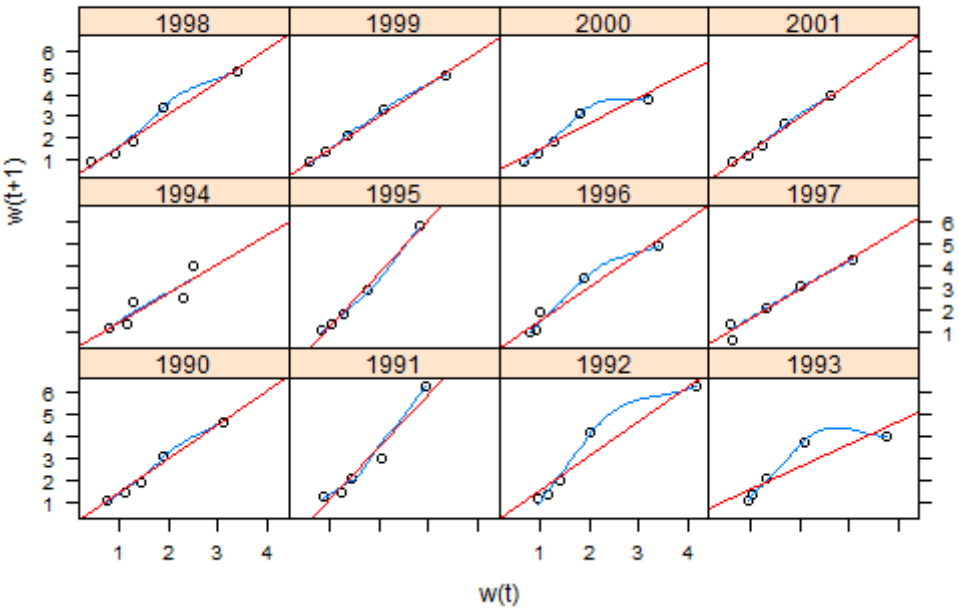



**Figure 11.6.3. Weight of cod at time (t+1) vs. weight at time t, following single cohorts wich are indicated by the year above each panel.**

### 11.6.2 Advice on community and foodweb indicators

Four community indicators were selected for presentation in the present version of multispecies advice: Natural mortality by age, percentage of total mortality caused by natural sources, the Large Fish Indicator and the biomass by guild. None of these suggested foodweb indicators have associated reference points yet as reference points for the Large Fish Indicator pertains only to survey catches and not assessment output. Hence, time-series of the different indicators are given which can be supplemented by the development in indicators relative to agreed reference points as these are defined.

#### Natural mortality by age

The development in predation mortality by age can be seen in Figure 11.6.4. The basic natural mortality from other sources (0.2) should be added to this to get the total natural mortality.

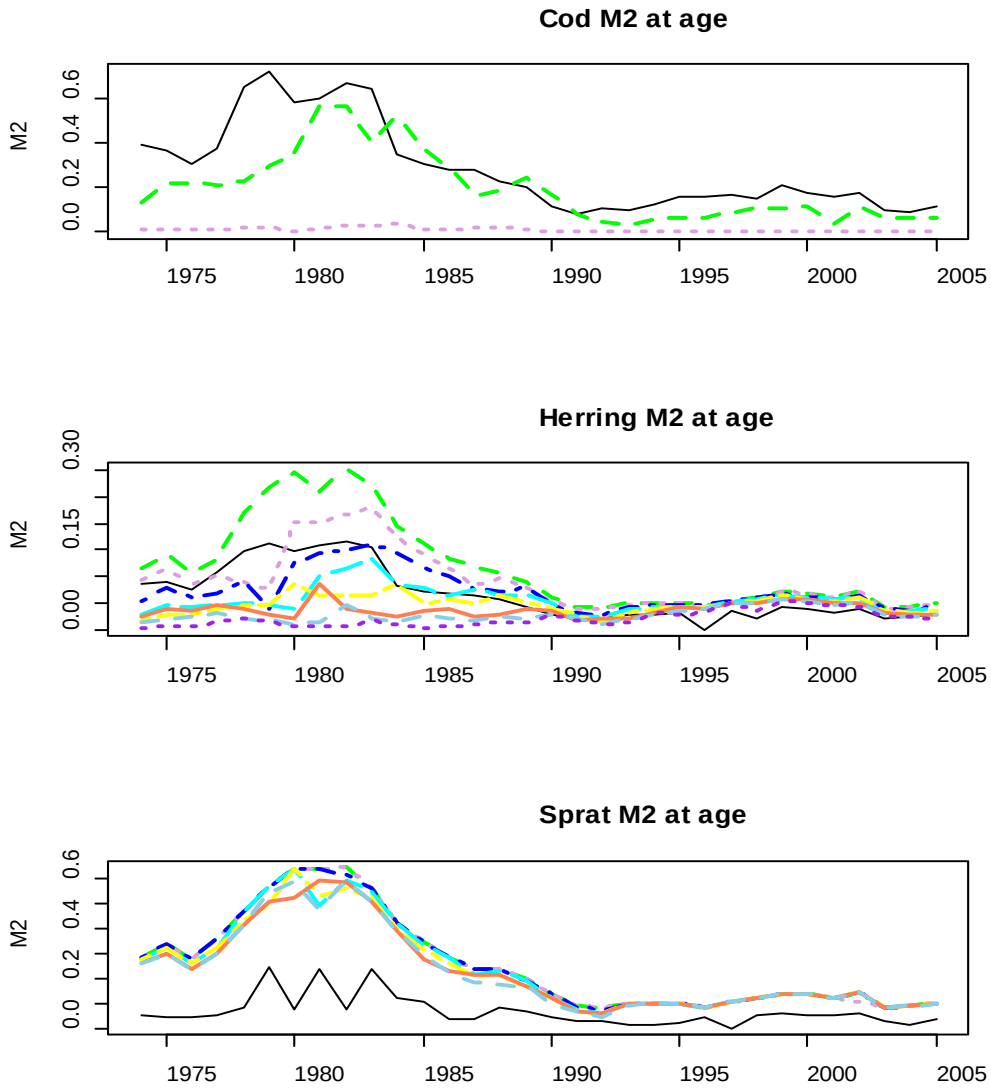



**Figure 11.6.4. Temporal development in natural mortality of age. Age 0: black, Age 1: red, Age 2: green, Age 3: blue, Age 4: cyan, Age 5: magenta, Age 6: yellow, Age 7: grey.**

### Percentage of total mortality caused by natural sources

With the exception of cod and sprat of ages 2+, predation mortality is more than 50% of the total mortality in the majority of the years (Figure 11.6.5).

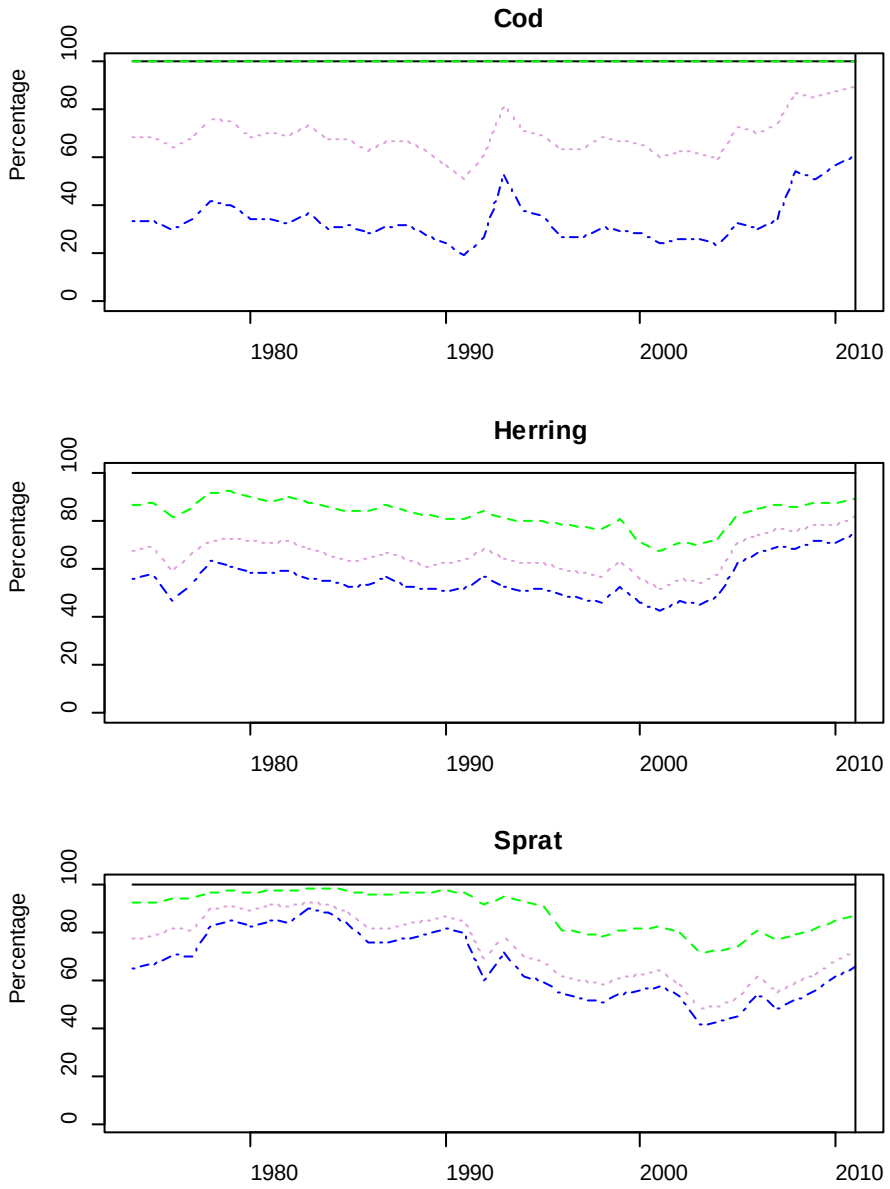



**Figure 11.6.5. Percentage of total mortality which is due to natural sources.**

### 11.6.3 Large Fish Indicator

The large fish indicator was estimated from assessment biomass at age assuming that all fish in a given age group have the average length of that age. This coarse assumption can be relaxed in future as length distributions of each age group are available. The only species included is cod.

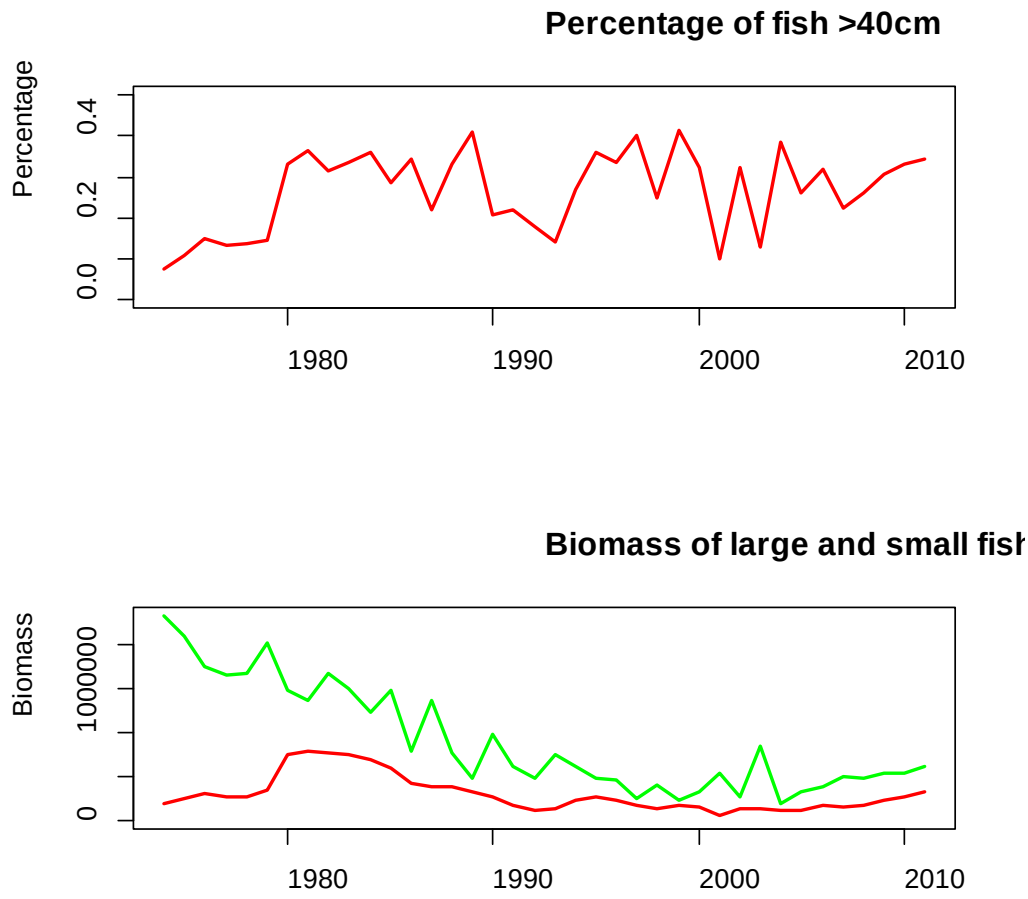



**Figure 11.6.6. Temporal development of the Large Fish Indicator estimated from assessment biomass at age. To demonstrate the causes of changes in the LFI, the biomass of small (green) and large (red) fish are given in the lower panel.**

### 11.6.4 Biomass by guild

Guilds were defined as forage fish (herring and sprat) and demersal fish (cod). The development in the biomass of each of these guilds are seen in Figure 11.6.7.

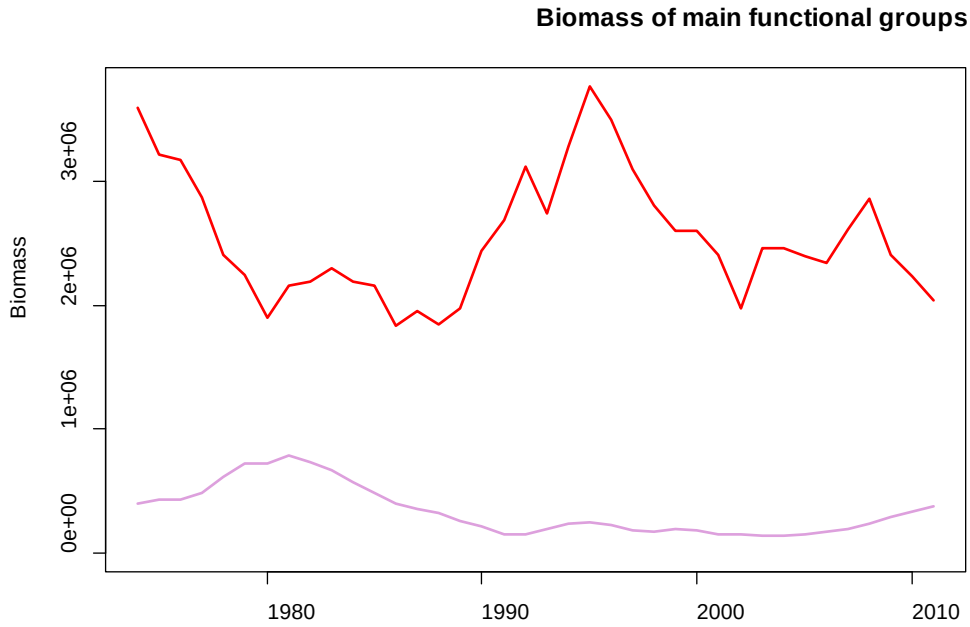



**Figure 11.6.7. Biomass of main functional groups (guilds). Red: pelagic, purple: demersal.**

#### 11.6.4.1 Data for download

Tables of natural mortality and predator biomass are available for download at www.ices.dk/reports/SSGSUE/2012/WGSAM/SMS_BS_summary.csv.

### 11.6.5 Advice on the combination of target fishing mortalities producing precautionary results in a multispecies environment, advice on the combination of target Fishing mortalities producing close-to-MSY in a multispecies environment and advice on ranges of F observed historically and hence defining the range in which confidence in the model is high

This work was initiated at WKMULTBAL in 2012. To proceed with precautionary results, it is necessary to define limit reference points for all three stocks. WGSAM suggests that WKBALT in 2013 define new reference points and that participants of WGSAM attend WKBALT to the complete the multispecies advice for the Baltic Sea based on a full stochastic simulation.

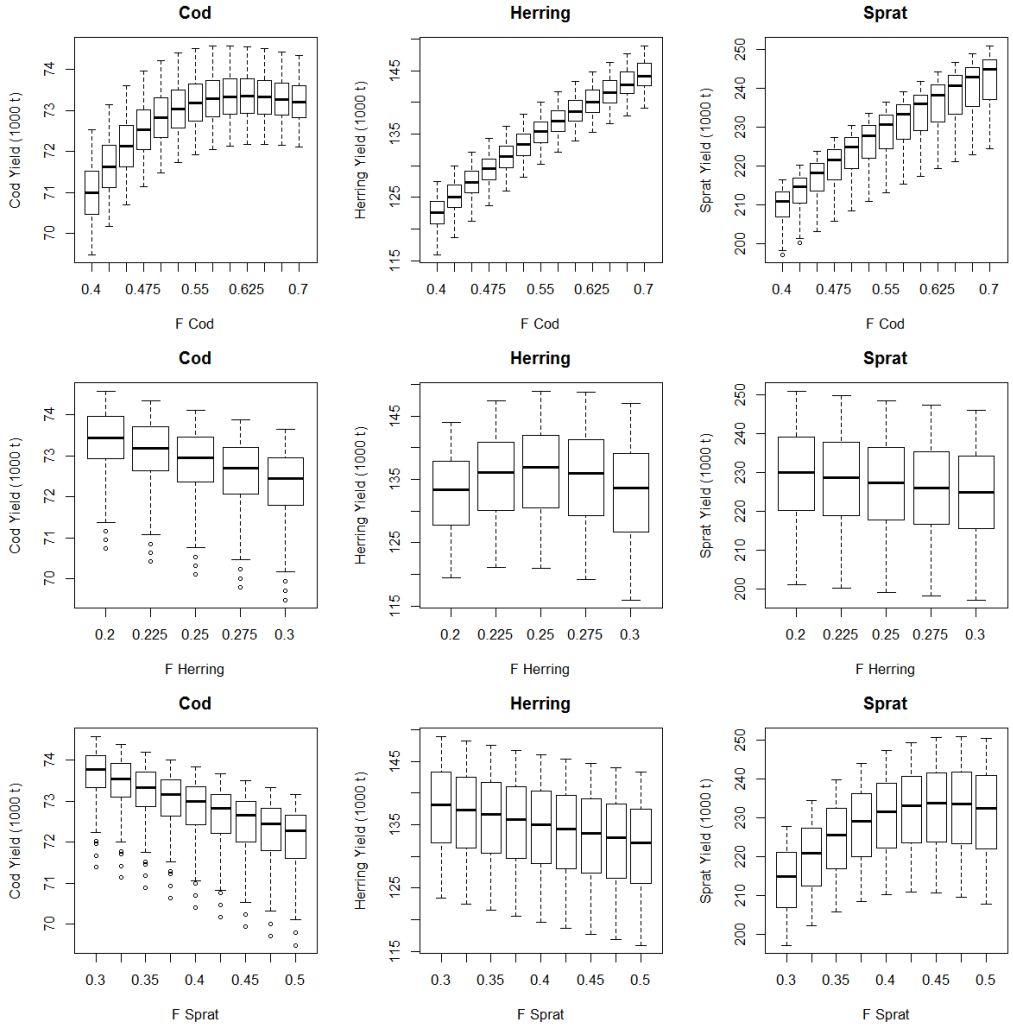



**Fig 11.6.5.1. Yield of (from left to right) cod, herring and sprat as a function of (top to bottom) target F on cod, herring and sprat.**

### 11.6.6 Advice on important interactions and trade-offs

Yields of virtually all species are strongly affected by the target F on cod and the effect of changes in predation on yield is as large as or larger than the effect of target F on herring. A high F on cod and saithe releases the two smaller gadoids from predation and increases both biomass and yield of them. For haddock yield remains virtually unrelated to haddock F. However, for the remaining species, there is a reasonable relationship between species-specific F and yield. All species have at least two strong off-diagonal interactions, and whiting and sandeel yield are strongly affected by F on three other species (saithe, cod and sandeel and saithe, cod and haddock, respectively). Norway pout and sprat are the only two species where F does not significantly affect yield of other species. Whiting and haddock fishing mortalities are the second least important for other species with only two interactions each.

**Table 11.6.1. Importance of fishing mortality (rows) on yield (columns). Dark shading indicates high importance, light indicates low importance. Criteria: Dark: Median, upper and lower whiskers change, medium grey: median, upper and lower whiskers change but change is small, white: no noticeable change in median or whiskers. White line indicates effects of species F on species yield.**

| F\Yield | Cod | Herring | Sprat |
|---|---|---|---|
| Cod | | | |
| Herring | | | |
| Sprat | | | |

### 11.6.7 Advice on any other relevant multispecies issues

No advice was given on this topic.

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# Annex 2: Agenda

| | |
|---|---|
| | **Monday 22-Oct** |
| 1300 | Opening of the meeting |
| | Adoption of ToR and Agenda |
| | Volunteers to work on different ToRs and overview of presentations prepared for the meeting |
| | ToR a, Presentations describing further progress in multispecies and ecosystem modelling |
| 1600 | Tea |
| | ToR a, Presentations describing further progress in multispecies and ecosystem modelling |
| | Develop Following Day Workplan, Updates from earlier in day, Wrap up |
| 1730 | Adjourn |
| | **Tuesday 23-Oct** |
| 900 | Initial presentations of ToR b: Key runs. |
| | Key runs scheduled for 2012: |
| | SMS Baltic Sea, Gadget Bay of Biscay? Others?<br>Update runs:? |
| 1030 | Coffee |
| | Disussion on ToR c, h and i, volunteers IDed to write up |
| 1300 | Lunch |
| 1400 | Reconvene |
| | Disussion on ToR e and f, possible twinning with WGFE. Volunteers IDed to write up |
| 1600 | Tea |
| | Continue discussion on ToR e and f, Subgroup reporting<br>Wrap up ToR a |
| | Develop Following Day Workplan, Updates from earlier in day, Wrap up |
| 1730 | Adjourn |
| | **Wednesday 24-Oct** |
| 900 | Revisit ToR b, Key run reports |
| | Revisit, as need be, ToR c, e, f, h and i |
| 1030 | Coffee |
| | ToR d and g, discussion and workplan |
| 1300 | Lunch |
| 1400 | Reconvene |
| | |
| 1600 | Tea |
| | Possible twinning with WGFE |
| | Subgroup reporting |
| | Develop Following Day Workplan, Updates from earlier in day, Wrap up |
| 1730 | Adjourn |
| | |
| | **Thursday 25-Oct** |
| 900 | Preliminary check on WG report elements |
| | Revisit, as need be, ToR a-i |
| 1030 | Coffee |
| 1300 | Lunch |
| 1400 | Reconvene |

| | |
|---|---|
| 1600 | Tea |
| | Possible twinning with WGFE |
| | Subgroup reporting |
| | Drafting session |
| | Develop Following Day Workplan, Updates from earlier in day, Wrap up |
| 1730 | Adjourn |
| | **Friday 26-Oct** |
| 900 | Revisit, as need be, ToR a-i |
| 1030 | Coffee |
| | Subgroup reporting |
| | Drafting session |
| | Scope out next year meeting plan, schedule, ToR |
| 1500 | Adjourn |

## Annex 3: WGSAM terms of reference for the next meeting

The **Working Group on Multispecies Assessment Methods** (WGSAM) chaired by Daniel Howell*, Norway and Steve Mackinson*, UK, will meet in Stockholm, Sweden, 21-25 October 2013 to:

Multiannual ToRS, where possible coordinated with other WGs

a ) Report on **further progress** and key updates in multispecies and ecosystem modelling throughout the ICES region;

b ) Report on the development of **key-runs** (standardized model runs updated with recent data, producing agreed output and agreed upon by WGSAM participants) of multispecies and eco-system models for different ICES regions (including the Baltic EwE 2013, Barents Sea 2014, North Sea EwE 2014, North Sea SMS 2014, Baltic Sea SMS 2015 and others as appropriate);

c ) Where possible, develop standards for 'Key Runs' of other modelling approaches (e.g. Size spectra, TGAMs);

d ) Develop and compare **foodweb and ecosystem indicators** (e.g. from the MSFD) and advice produced by multispecies key runs (preferably together with WGFE and WGECO);

e ) Report on progress on including **new stomach samples** in the ICES area in multispecies models;

f ) Explore the consequence of multispecies interactions and environmental factors in practical multispecies advice for fisheries management (**MSY related and other biological reference points**);

g ) Compare methods used to include **spatial structure** (predator prey overlap) in multispecies prediction models (preferably together with WGIPEM);

h ) Work towards providing ecosystem advice consistent with species and technical interaction in mixed fisheries (preferably together with WGMIXFISH);

There have been suggestions that WGSAM should meet right after spring assessment groups to provide annual updates of multispecies advice. As the group currently focuses on methods and long-term advice, the group considered the current season to be preferable to moving to autumn.

**Connection with the ICES science plan**

The work outlined above fits well with the high priority research topics given in the ICES Science Plan for 2009-2013, and apply to all three thematic areas (Understanding ecosystem functioning, Understanding interactions of human activities with ecosystems, and Development of options for sustainable use of ecosystems).

**Supporting information**

| | |
|---|---|
| Priority: | Multispecies assessment modelling is essential to the development of viable long-term management strategies. |
| Scientific Justification and relation to action plan: | The increased emphasis on ecosystem management (e.g. under the revised Common Fisheries Policy), and a move away from advising on single-stocks in isolation, necessitate consideration of interactions between key fish stocks and the ecosystems of which they are part. |

| | |
|---|---|
| | Historically the various ICES multispecies working and study groups have acted as a useful conduit, drawing together advice and quantitative outputs from many different assessment groups and combining these into an integrated product of direct use to managers and researchers. The past several meetings of WGSAM showed that there is much ongoing work within this field, and that there is a need for a pan-European (and more fully, northern hemisphere) forum for reviewing progress, and for learning about the 'best practice' of other research groups (ongoing ToR a).<br><br>Multispecies models are used to provide updates of natural mortality M for inclusion in conventional single-species stock assessments of several stocks (ToR b). Consequently, it is considered useful to have occasional 'key-runs' for each region, whereby time-series are updated and model configurations are agreed and 'peer reviewed' by a number of regional experts. WGSAM will continue to work towards improved key-runs in the Barents Sea, Bay of Biscay, Baltic and North Sea, as well as working towards significant improvements in model functionality, for example the development and application of new model types such as production models and size spectra models and the inclusion of new stomach data (ToRs c and e).<br><br>The ICES Science Plan for 2009-2013 highlights a top research priority for better understanding the impacts of fishing on marine ecosystems, in particulas on the biodiversity and helath of ecosystems. In 2013-2015, WGSAM will continue their efforts in the development of foodweb and ecosystem indicators (ToR d), including both existing and new potential indicators in support of this top research priority. The work on investigating the effect of species interactions and environmental effects on biological reference points including those related to MSY will continue to support long-term management considerations (ToR f).<br><br>Spatial structure is an important factor affecting species interaction but the knowledge of how this is included in models and how spatial structure is likely to develop over time in forecasts requires further work by WGSAM in cooperation with other ICES Working Groups addressing spatial issues (ToR g). The delivery of multispecies medium term advice which is consistent with practically implementable management requires a coupling between interactions during the fishing process and biological interactions. Work on this will be initiated under ToR h.<br><br>Other priority research areas that have been highlighted in the ICES Science Plan and which will be addressed by WGSAM at its 2012 meeting include: impacts of fishing on marine ecosystems (ToRs b, c, d and g), and marine living resource management tools (ToRs a, b, d, f, g and h). |
| Resource Requirements: | – |
| Participants: | Approx 20. Expertise in ecosystem, modelling and fish stock assessment from across the whole ICES region. |
| Secretariat Facilities: | None |
| Financial: | No financial implications |
| Linkage to Advisory Committees: | ACOM |
| Linkage to other Comities or groups: | WGDIM, WGBIFS, IBTSWG, WGECO,WGMIXFISH, WGFE, WGINOSE, WGAIB, WGNARS, WGIPEM, most assessment Expert Groups, most EGs in the regional Seas Programme |
| Linkages to other organizations: | – |

# Annex 4: Recommendations

| Recommendation | For follow up by: |
|---|---|
| 1.Evaluate the appropriateness of the proposed foodweb and ecosystem indicators listed in Table 7.1.1 in the report of WGSAM 2012 and consider a joint future ToR with WGSAM to continue this work. | WGECO |
| 2.Consider the introduction of a joint future ToR with WGSAM to develop the work towards providing ecosystem advice consistent with species and technical interaction in mixed fisheries. | WGMIXFISH |
| 3.Consider the introduction of a joint future ToR with WGSAM to work on including spatial structure (predator prey overlap) in multispecies prediction models | WGIPEM |
| 4.Evaluate the proposed multispecies advice format for the Baltic Sea and produce advice on precautionary reference points to allow the estimation of precautionary FMSY values. | WKBALT |

## Requests to other groups:

WGECO: Evaluate the appropriateness of the proposed foodweb and ecosystem indicators listed in Table 7.1.1 in the report of WGSAM 2012 and consider a joint future ToR with WGSAM to continue this work.

### Explanation

The work on the development of foodweb and ecosystem indicators should continue in cooperation between the two groups, utilizing the knowledge of indicator evaluation in WGECO and the foodweb modelling expertise in WGSAM.

WGMIXFISH: Consider the introduction of a joint future ToR with WGSAM to develop the work towards providing ecosystem advice consistent with species and technical interaction in mixed fisheries.

### Explanation

There is currently considerable interest in providing multispecies advice which is consistent with both biological and technical interactions. This work should continue in cooperation between WGMIXFISH and WGSAM, initially focusing on providing medium term advice.

WGIPEM: Consider the introduction of a joint future ToR with WGSAM to work on including spatial structure (predator prey overlap) in multispecies prediction models

### Explanation

The work on spatial modelling of species interactions requires knowledge of both spatially explicit abundance of fish and the processes that shape their movements. Spatially explicit forecast models rely on estimates of fish movement, an area which is not currently sufficiently developed to allow estimates of FMSY when species overlap varies over time.

WKBALT: Evaluate the proposed multispecies advice format for the Baltic Sea and produce advice on precautionary reference points to allow the estimation of precautionary FMSY values.

**Explanation**

The WKBALT will be the first multispecies benchmark and provides an excellent platform for introducing the multispecies advice into assessment working group advice. Further, the advice on precautionary in a multispecies context cannot be continued without the definition of suitable reference point, which is one of the ToRs of WKBALT.