Deliverable No. 4.6

Project acronym:
MareFrame

Project title:
"Co-creating Ecosystem-based Fisheries Management Solutions"
Grant agreement No: **613571**
Project co-funded by the European Commission within the
Seventh Framework Programme
Start date of project: **1ˢᵗ January 2014**
Duration: **48 months**

Due date of deliverable:	31/11/2014
Submission date:	31/12/2016
File Name:	D4.6 MAREFRAME_Parameterisation part 4.Alternative model run for each case study which replicates the time series of the commercial fish species, GES, economic and social (EAFM) indicators
Revision number:	01
Document status:	Final[1]
Dissemination Level:	PU[2]

Revision Control

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[1] Document will be a draft until it was approved by the coordinator
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Deliverable D4.6

Parameterisation 4. Alternative model run for each case study which replicates the time series of the commercial fish species, GES, economic and social (EAFM) indicators

December 2016

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This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no. 613571

EXECUTIVE SUMMARY

This report, deliverabele 4.6 (D4.6), is a deliverable of Work Package 4 (WP4 – Ecosystem models and assessment models) of the FP7 MareFrame research project. The aim of the MareFrame project is to identify management strategies which will achieve Good Environmental Status (GES) by applying a minimum of two ecosystem models on each of eight different case studies across Europe. These eight case studies are: West of Scotland, the Baltic Sea, Iceland, the Strait of Sicily, the North Sea, South western waters, the Chatham rise, and the Black Sea. This report describes the (main) alternative ecosystem models that have been applied to each of the Mareframe case studies. All of the case studies have tested at least two ecosystem models and some have tested a variety of different models, selecting a primary and a secondary from the list. The primary model chosen in each case has been reported in other deliverables of WP4. During the project planning phase it was envisaged that the alternative model would be set up to also produce all the same outputs as the primary model. During the model development and testing phase more important issues and considerations emerged, both scientifically and in relation to the co-creation process with stakeholders. These new considerations include:

- support for a model framework reduced so the focus was instead placed on newer methods of statistical estimation (eg WoS, Gadget)
- results from more comprehensive model becoming available sooner than expected (eg Atlantis in Icelandic waters)
- increased emphasis on knowledge transfer rather than a complex model applied to a data-poor situation (Black Sea)
- overwhelming support for a much wider application of a single model (Green model, North Sea and others)
- developments in methods to synthesis outputs from ecosystem models (through the Green model, originally developed for the NS only)

The results show point estimation and, where available, uncertainty estimates, measures of GES and economic indicators. Additionally, comparisons between different ecosystem models are provided in a some cases, though this is more a task for other deliverables such as D7.2 and D5.3.

1. INTRODUCTION

Marine fisheries are a resource of political, economic and social importance in the European Union and in some case have a significant contribution towards the Member States' economy. It is therefore essential to protect the European marine environment in order to maintain its health and ensure sustainable production from fish stocks in the future. To achieve this, the Ecosystem Approach to Fisheries Management (EAFM) has been proposed at various international summits and conferences as the best means by which sustainable exploitation can be achieved. The MareFrame EU research project (http://www.mareframe-fp7.org/) aims to apply and test the EAFM on eight worldwide case studies which cover a variety of ecosystems, fisheries-related issues, and data availability. These eight case studies are: West of Scotland, the Baltic Sea, Iceland, the Strait of Sicily, the North Sea, South western waters, the Chatham rise of New Zealand, and the Black Sea. Ecosystem models will be employed in order to identify appropriate management strategies on a multispecies & ecosystem basis for a variety of objectives as determined in the co-creation approach of the project.

Contrary to the traditional monitoring of ecosystem indicators based upon observations, the tremendous advantage of the MareFrame approach to dealing with EAFM lies in performing forward simulations of any pressure applied to the ecosystem in order to forecast what will be the associated impact on ecosystem indicators, and ultimately, the ecosystem status. It is, therefore, crucial to ensure that ecosystem models employed in MareFrame are parameterised to provide a reliable representation of the ecosystem. To assess whether an ecosystem model is correctly parameterised, the outputs of the model for a historical period are compared to data observed over that period. If the model outputs match the historical trend then it can be inferred that the model provides a reliable representation of the past reality which is the best foundation to predict the future. Within the MareFrame project, a minimum of two ecosystems models are applied in each case study in order to assess the robustness of EAFM despite the type of model employed. This report describes the second ecosystem model to be employed in each case study.

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2. WEST OF SCOTLAND CASE STUDY: GADGET

2.1. Introduction

The West of Scotland Ecosystem comprises the shelf area west of Scotland (ICES subarea VIa) and supports several valuable fisheries: (i) a demersal mixed fishery targeting mainly cod, haddock, whiting, European hake, saithe and monkfish, (ii)a shellfish fishery targeting the Norway lobster and (iii) a pelagic fishery targeting mainly Atlantic mackerel, horse mackerel, herring and blue whiting. These fisheries are currently managed through TACs and quotas set each year individually for each stock without multispecies considerations. Additional measures such as effort and gear restrictions and closed areas are also in place (for full CS description see D 5.1).

The West of Scotland fisheries currently face several management issues. Firstly, the stocks of cod and whiting are currently depleted well beyond precautionary levels. Secondly the population of grey seals has been increasing over the past 2 decades and has been linked to an increase of predation mortality on cod which could jeopardise effort to recover the stock (Cook and Trijoulet, 2015). In addition, the presence of 2 depleted stocks in a mixed fishery is likely to result in choke species which will jeopardise the productivity fishery when the landings obligation comes into place in 2019. Under the MSFD, GES must be achieved by 2020. This includes bringing all exploited stocks above precautionary levels. While not all descriptors can be assessed in the a fisheries context, an ecosystem approach allowing for multispecies consideration and ecosystem indicators must be employed to identified the best management alternatives.

The first model employed in the west of Scotland case study is Ecopath with Ecosim (EwE). EwE allows for the inclusion of a large number of species (41 functional groups in our case) covering the trophic levels of the entire foodweb and therefore is a useful tool to assess prey-predator interactions as well as the overall ecosystem health. However it is not length- or age-based and predcludes investigating technical management measures such as gear selectivity, and the lack of distinction between bycatches and discards prevents investigating discards-related issues (e.g. landings obligation). To address these shortcomings, the second model employed in the west of Scotland case study is Gadget: an age- and length- based multispecies assessment model. While Gadget is more data and computation demanding to to its higher life-stages resolution and thus can only include a few species, it provides a more realistic simulation of biological processes. The combination of these two modelling tools complementing each other, EwE and Gadget, will allow for a thorough exploration of alternative management scenarios.

2.2. Methods

2.2.1. Multispecies model

Gadget is a statistical modelling environment which can be used to describe the population dynamics of species within a given ecosystem. Models can be implemented on a single species basis (i.e. similar to typical stock assessment models), or in a multispecies framework where several species interact through predator-prey relationships. Multiple, interacting fishing fleets may also be specified in Gadget models, with each fleet removing predetermined target species. In addition, discards and misreported catch may be modelled. Gadget has the flexibility to incorporate data from many different sources to a single model, including commercial catch-at-age data, scientific survey indices, and stomach content data. Models are fitted to the data using a formal statistical approach, whereby each species' population is simulated given a set of initial conditions, population parameters (e.g. recruitment, fishing mortality) are estimated by maximum likelihood, and the simulated population is compared to the data through negative log-likelihood functions. The fitting procedure itself is objective, but care must be taken to ensure the structure of the model is ecologically sensible. Once a satisfactory model is attained, Gadget can provide forward projections of various population indices based on the parameters of the modelled ecosystem.

Since Gadget can only include a few species, the three following species were chosen: cod, haddock and whiting. The choice of cod and whiting was evident: these are the two stocks currently depleted in west Scotland and one of the objectives of MareFrame is to identify the management strategy which would allow these two stocks to recover. Haddock was chosen as it constitutes a major economically important stock for Scottish fisheries. In addition, these three stocks have all been studied extensively compared to other species and as a result the biological data needed to parameterise Gadget is readily available. These species predate each other at different stages in their life which the length/age dimension of Gadget allows modelling. Another species was chosen to be added to the model: grey seals. The potential negative impact of grey seal predation on gadoid stocks is mentioned in the literature (Alexander et al., 2015), and it has recently been proposed that the increased grey seals predation could be hampering the recovery of cod (Cook et al., 2015). The addition of grey seals to the model will allow exploring the length-based predation reported by Cook et al. (2015).

The following data needed to parameterise Gadget was gathered for all species:
- Data: commercial landings & discards (numbers-at-age), survey indices & survey age/length distributions
- Three commercial fleets for each model: landings, discards, misreporting (1995-2005)
- Survey fleets operating in Q1 and Q4 (selectivity specified using the time variable functionality in gadget to reflect changes in survey gear from 2011 onwards)

In order to achieve a multispecies Gadget model the following approach was followed:

- First, single species models were built and parameterised independently for cod, haddock, whiting and grey seals
- Secondly, the four single species models were connected to one another vioa prey-predator interactions i.e. replacing natural mortality by the appropriate predation mortalities to achieve a multispecies model.

2.2.2. Indicators

Ecosystem indicators

Whilst Gadget is a multispecies model, it cannot be considered an ecosystem model *per se* since it only includes a few selected species which represent only a fraction of the species and trophic levels present in the foodweb. Therefore, it is not possible to derive foodweb ecosystem indicators such as mean trophic index, pelagic to demersal ratio, etc. from our Gadget multispecies model. However cod and haddock both have biological reference points defined (although for a larger stock area for haddock), whilst whiting used to have biological reference points defined (see Alexander et al., 2015) which can be employed as best available estimates. As a result, the biomass of mature individuals (i.e. individuals over the length/age at 50% of maturity) can be used as an indicator of stock status and the biological reference points (Blim, Bpa, Flim, Fpa) can be used to assess the status of these stocks and whether or not there are within safe limits, as per defined by Good Environmental Status.

Economic indicators

Economic data was obtained from the Scientific, Technical and Economic Committee for Fisheries *(*STECF*) of the European* Commission at https://stecf.jrc.ec.europa.eu/data-reports. While landings data are available per ICES area, the economic data per fleet given in STECF are only available for the whole northeast Atlantic (FAO area 27). In order to obtain estimates for our model area, the economic data from STECF was scaled down using the proportion of landings made in area 6a compared to the landings made in area 27. Once the data was scaled down, the following was extracted for each species considered in Gadget (i.e. cod, haddock and whiting):

- Historical profit
- Historical cost, as the sum of crew costs, unpaid labour costs, energy costs, repair costs and other variable costs
- Price per species by dividing the revenues by the price

It was assumed that cod, haddock, and whiting were caught only by demersal trawl and only data form these fleet/gear were considered. While historical costs are known, future costs over the simulation period are unknown. In order to estimate future costs in fishing scenario

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simulation, costs coefficients were calculated to relate costs to fishing mortality for demersal species following the work form Quaas et al. (2012) as follows:

$$Cost\ coefficient_{species} = Cost_{demersal\ trawl,species} \Big/ Fishing\ mortality_{species}$$

Using the landings returned by the model, the price per species and the cost coefficients as described above, the profit was then calculated for each species as follows:

$$Profit_{species} = \big(landings_{species} * price_{species}\big) - \big(cost\ coefficient_{species} \\ * Fishing\ mortality_{species}\big)$$

2.3. Results

The results from the single species Gadget models for cod haddock and whiting are displayed in figures 3.1, 3.2 and 3.3.
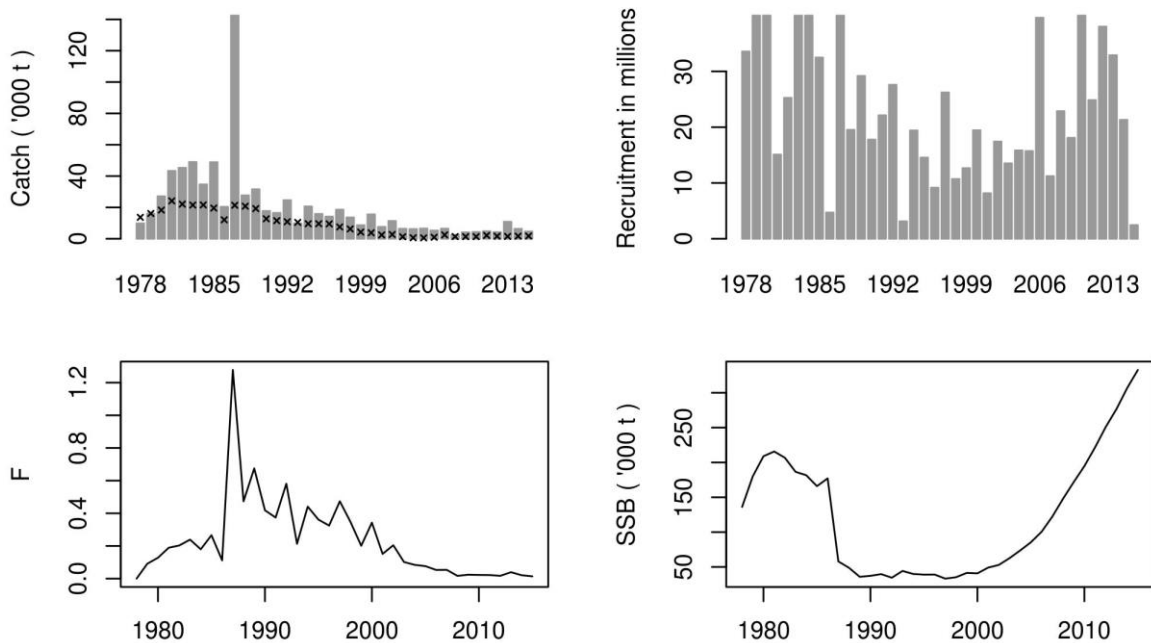
Figure 3.1. Outputs from the single species Gadget model for cod. Black crosses on the catches histogram are observed values.

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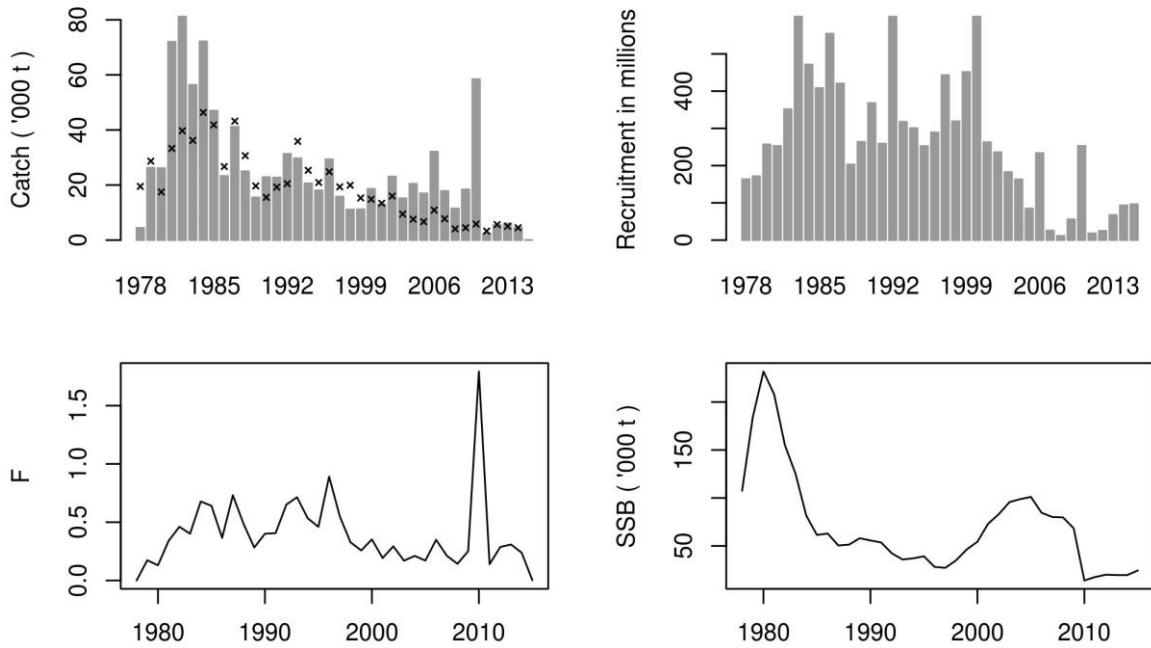
Figure 3.2. Outputs from the single species Gadget model for haddock. Black crosses on the catches histogram are observed values.
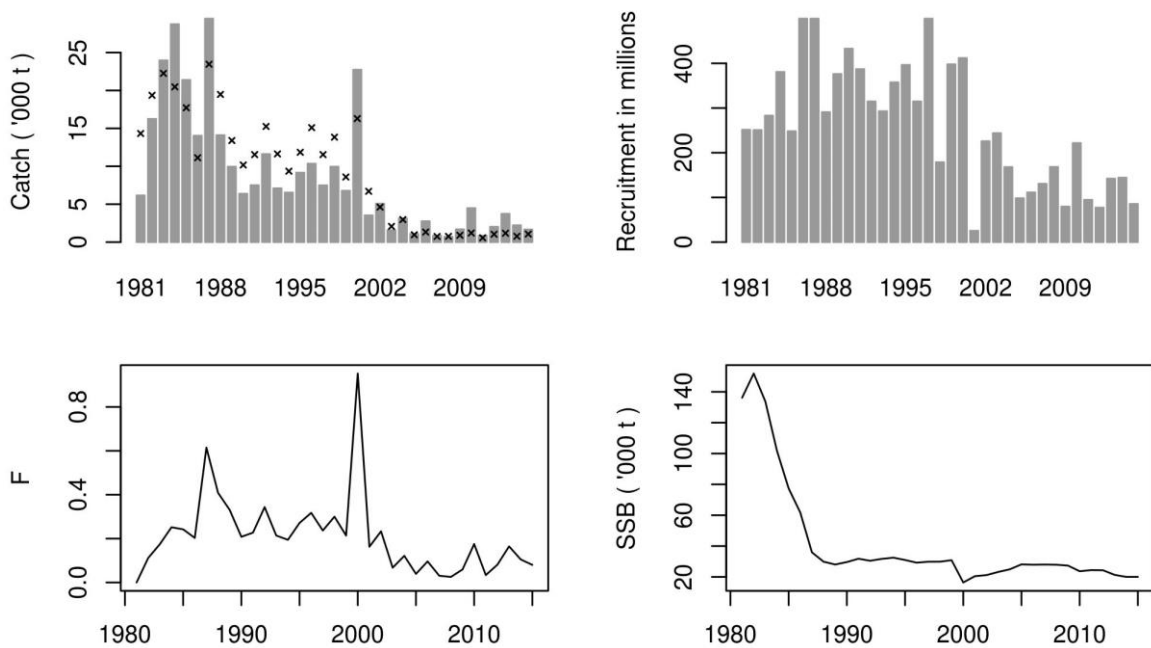
Figure 3.3. Outputs from the single species Gadget model for whiting. Black crosses on the catches histogram are observed values.

Both the haddock and the whiting single species Gadget models return outputs similar the ones from the current TSA stock assessment models, indicating a satisfactory parameterisation on a

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single species basis. For both species, Gadget is able to replicate the historical catch time series fairly well, and the trend in biomass matches that of the stock assessment.

For cod on the other hand, while catches are replicated well by Gadget the biomass shows a dramatic increase in the latter years which contrast with the depleted biomass currently reported by the TSA stock assessment which indicates that Gadget is not suitably parameterised for cod. This discrepancy may be due to misreported catches as suggested by Cook et al. (2015): the misreported catches would actually be less than reported in the stock assessment while the corresponding mortality would in fact be due to seal predation. Therefore it is hoped that adding seal predation to the model will result in a satisfactory parameterisation for cod.

The next steps for the multispecies Gadget model for the west coast will be to:

- Refine the parameterisation of the cod single species model
- Develop the seal model for which new seal diet data, needed to parameterise the model, is expected shortly
- Combine the single species models into a multispecies model to explore management alternatives currently being investigating using EwE

2.4. References

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3. BALTIC SEA CASE STUDY: GADGET

3.1. Introduction

The case study approaches the cod, herring, and sprat fisheries in the ecosystem context in the Central Baltic. Rather good biological knowledge exists about these species, their trophic interactions and population dynamics. The Baltic Sea pelagic fish biomass is dominated by herring and sprat. Herring is one of the key species due to its high abundance and role as a consumer in the pelagic food web, and as forage for cod, salmon and seal. Sprat competes for food with herring, and herring growth is considerably lower at high sprat densities than at low sprat levels. Sprat predates cod egg and larvae.

The general goal of the Central Baltic fisheries management and management objective as formulated by together with the case study stakeholders (17th December 2015) is "... the management of cod, herring, and sprat fisheries considering trophic interaction among the stocks in the Central Baltic, as well as the major environmental drivers influencing the dynamics of the harvested populations. The social and economic benefits that can be derived from these fisheries are explicitly acknowledged in the management process. Management recognises the small scale fishery targeting cod with gill nets as this segment provides employment opportunities for the coastal communities in particular. Sustainable harvesting of the major commercial stocks is the primary management interest and both spawning biomass and size structure of large predatory species should contribute to a resilient state. This will also indicate good status of a relevant biodiversity component preserving ecosystem functioning".

Developing multispecies long-term management plans for the pelagic ecosystem of the central Baltic is recognised as a priority towards an ecosystem approach to fisheries.

3.2. Methods

3.2.1. Multispecies model

The implementation of this type of Gadget multi-species models requires a number of intermediate steps, including the parametrization of single species models which are later linked by trophic interactions and/or interaction with fisheries targeting more than one species.

Both single and multispecies implementations are fitted to multiple datasets to estimate the model parameters. For each data component a specific likelihood function is used to compare the model output to the data during the estimation. Dataset and associated likelihood function are also referred hereafter as likelihood components.

The procedure of weighting different dataset follows the method proposed by (Stefansson 2003) and described in a protocol by Taylor et al. (2007). Implementation of this approach is done in Rgadget (Elvarsson 2010) which has been used for the iterative reweighting and model

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optimization. In essence, the procedure aims at objectively assigning weights to the different data components by evaluation of the individual fitting of each component. This is achieved by heavily weighting a likelihood component and by running an optimisation to minimise the negative log-likelihood function. The estimated negative log-likelihood for this component is taken as a measure of how well the model can best fit that dataset, and is divided by the number of degrees of freedom (approx. given by the discrete #Nyears, #Ntimesteps, #NageGroups, #NlengthGroups) of that component. This quantity is used as a variance estimate and its inversion is used as final weight for the likelihood component. The procedure is applied iteratively for each likelihood component until all datasets are weighted (for more details see Taylor et al. (2007)).

The three stocks are built around a similar quarterly based conceptual model with fishing and natural mortality occurring in all time steps, recruitment once a year in a specified quarter and one or more surveys occurring in different times of the year.
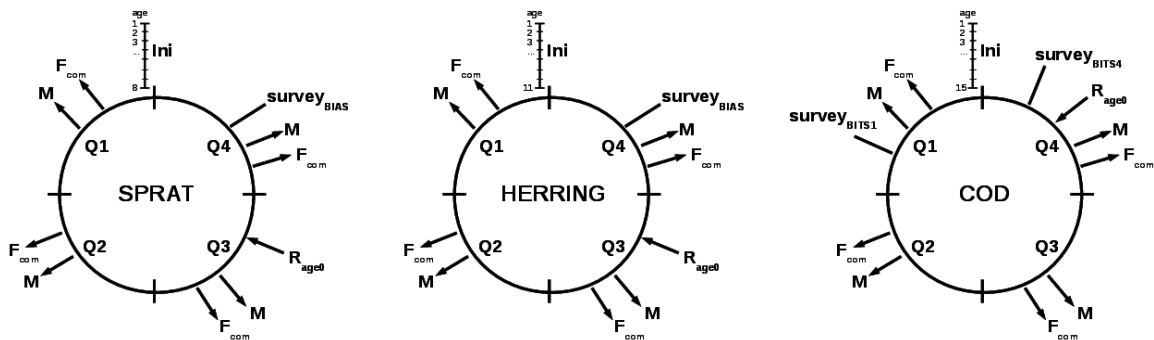
A more detailed description of model parametrization is presented in D5.3.

3.2.2. Indicators

GES indicators

The following GES indicators were calculated for cod, herring and sprat from the model output:*

- spawning stock biomass (SSB), related to the reproductive capacity of the stock [D3.2.1]
- fishing mortality (F), related to the level of pressure of fishing activity [D3.1.1]
- demersal-pelagic ratio (D/P), ratio cod/(sprat+herring) related to the ecosystem structure and relative proportion of ecosystem components [D1.7]
- proportion of mature fish (prop_mat), proportion of mature cod (or herring or sprat) in the stock, related to the population size distribution [D3.3.1] and relative reproductive capacity
- 95% of the length distribution (Len_95), 95 percentile of the length distribution in the cod stock, related to the population size distribution [adapted from D3.3.3]

* in squared brackets the corresponding MSFD descriptor or indicator

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Economic indicators

There are three fleet segments of interest defined in the case study, for which socio-economic indicators are calculated: active gears (bottom trawls) targeting cod, passive gears (gillnets, mostly small-scale fishery) targeting cod and pelagic fishery targeting sprat and herring. As a result of multiple consultations with stakeholders, we decided to calculate *Profit* and *Year-to-Year Similarity* of profit as economic indicators.

Publicly available economic data (2016-07_STECF_EU Fleet Economic data_fs level) and same assumptions used for the primary model EwE were applied here (see D4.5 for more details).

Profit was calculated as the earning before interests, taxes depreciation and amortization. For a given year: profit= landings*price – cost

Based on the work by Quaas et al. (2012) and Voss et al. (2014) we assumed costs in the cod fishery to depend on fishing mortality caused by the fishery, and in schooling fisheries (clupeids) on landed weight. The price per kg of cod payed to the gillnet fishery was corrected in relation to seal abundance, to represent damage of the catch (see D4.5 for details).

Year-to-Year Similarity expresses how much profit in one year is indicative of profits in the next year, which is useful e.g. for planning investments. It is calculated as the first-order autoregression in the time series of profit, i.e. how much one year is correlated with the next year. The indicator value is going to be higher if the time series has a monotonous trend (it is easy to predict next year's profit from this year's profit), and low if the time series shows both increases and decreases among years.

3.3. Results
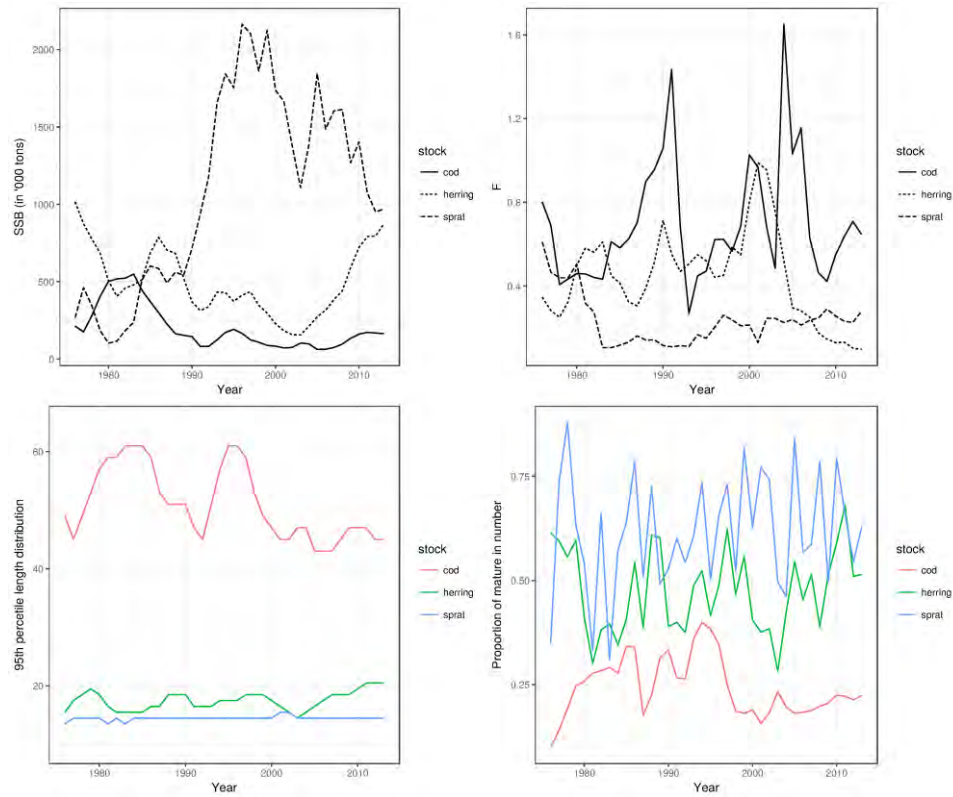
Figure 1. Time series of SSB, F, 95% of the length distribution and proportion of mature fish for cod, herring and sprat estimated by Gadget for the time period 1974-2013.
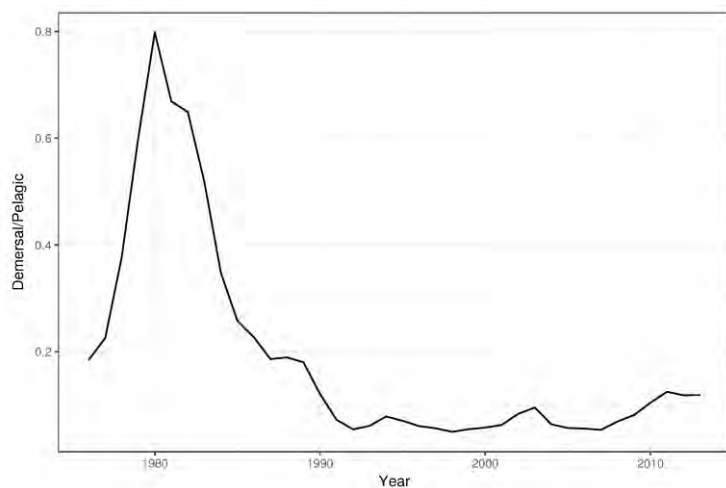
Figure 2. Time series of D/P ratio estimated with Gadget for the time period 1974-2013.
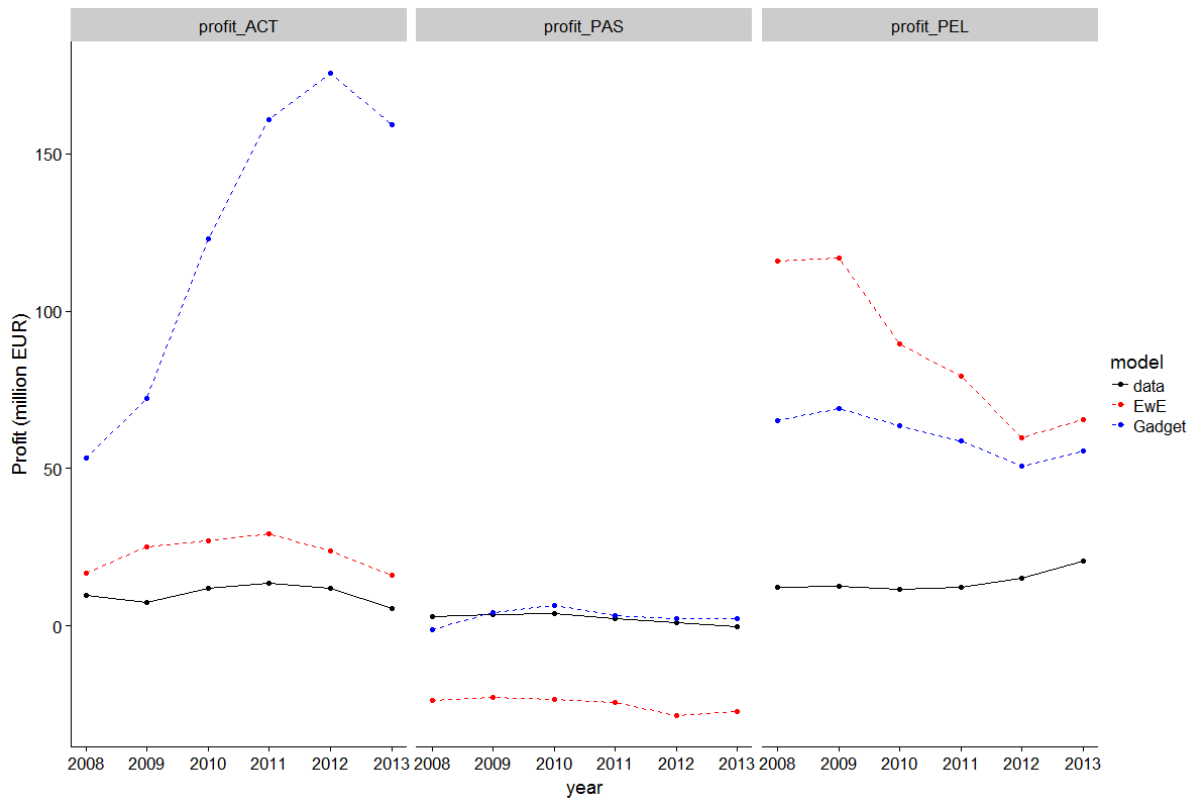
Figure 3. Comparison of the time series of profits estimated by Gadget (and EwE from D4.5) to data 2004-2013.
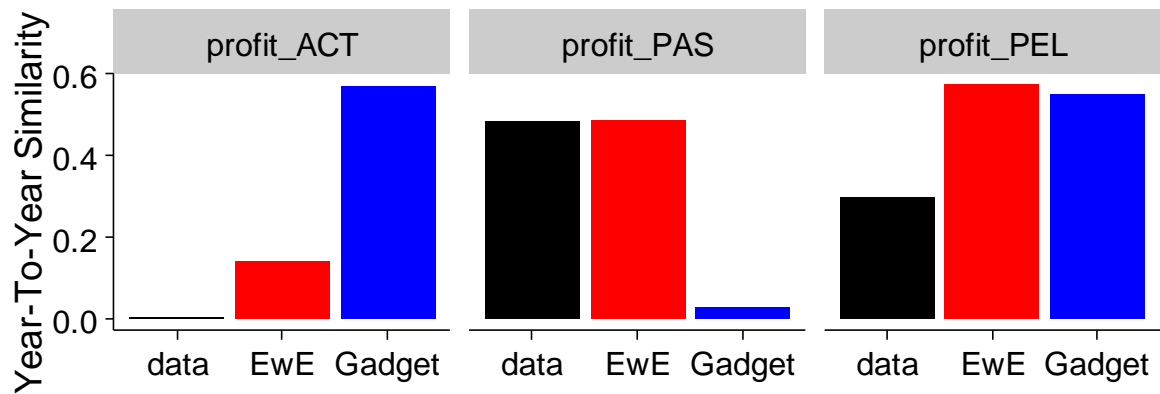
Figure 4. Comparison of Year-to-Year similarity of profit estimated by Gadget (and EwE from D4.5) to data 2004-2013.

3.4. References

Elvarsson B.T. 2016. Rgadget: R package for Gadget models.

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4. ICELANDIC WATERS CASE STUDY: ATLANTIS

4.1. Introduction

The Atlantis model (Fulton et al., 2004) was used as an alternative model for the Icelandic case study. It is a whole-of-an-ecosystem model that considers physical, chemical, biological and human components. The physical model includes the oceanography, i.e. the flow of water in the modelled area, temperature and salinity. The flow of water controls the advection of nutrients and plankton. Temperature and salinity have an effect on the cycling of nutrients and growth of flora and fauna within the model. The biology model contains the functional groups, their consumption and predation, growth and reproduction, movements and migrations. Human activities are modelled with a fisheries model. Groups with commercial values are harvested and the harvest rate is allowed to change between years which impacts the stock dynamics which consequently affects the total catches and economic profit.
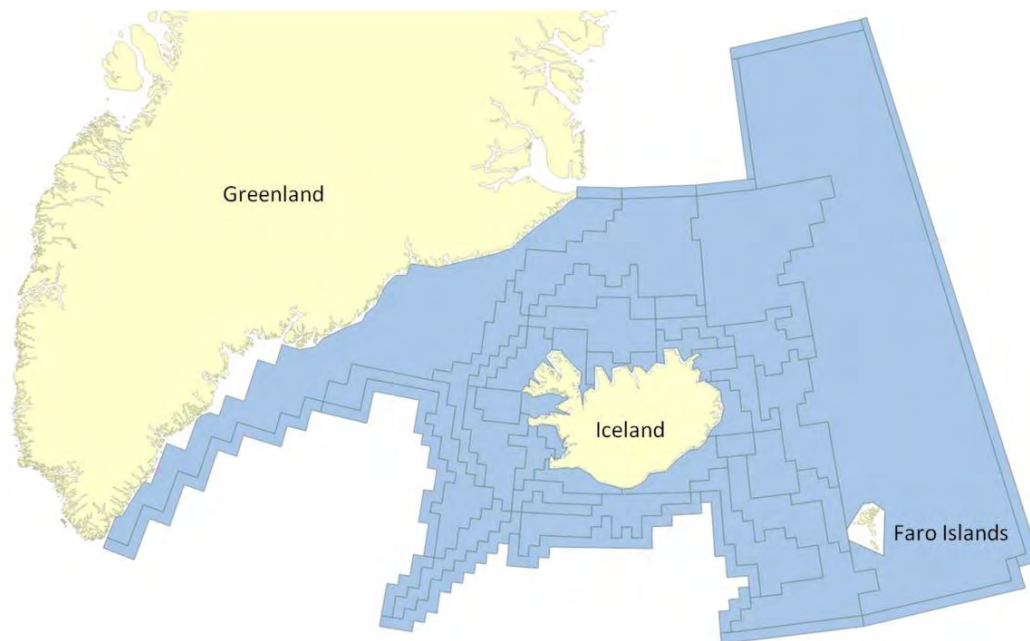
In this section the Atlantis model for Icelandic waters will be described. Simulated time-series of total biomass and catches of the commercial groups from the model will be compared with estimated biomass from stock assessments and landings data. Good environmental status (GES) indicators along with economic and social indicators will be calculated for the simulated period.

4.2. Methods

4.2.1. Model Structure

The oceanography model
The modelled area is 1,600,000 km^2 and covers the area from Greenland to Iceland and to the Faro Islands (Figure 1). The area has been divided into 53 boxes based on work done by Stefánsson et al. (1997). Active boxes where the actual biology is modelled are 36, 15 boxes are boundary boxes to make the oceanography work and 2 are islands (Iceland and Faro Islands). Each box is further divided into layers which depend on the depth of the box. The boxes have one sediment layer and can have a maximum of six water column layers (0-50m, 50-150m, 150-300m, 300-600m, 600-1000m and 1000m+). The oceanography data were taken from a hydrodynamic model (Logemann, 2013) and water fluxes, temperature and salinity were calculated for each box and layer each day from 1948 to 2012. A full model run is therefore 65 years and the time step is 12 hours. It takes 13 hours to run a full simulation of the model.

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Figure 1. The modelled area of the Icelandic Atlantis model.

The biology model

Functional groups

There are 52 functional groups in the model: 16 fish groups where 7 are at a species level, 3 groups of sharks and skates, 5 groups of mammals, 1 seabird group, 16 invertebrates, 6 primary producers, 2 bacteria and 3 detritus groups (

Table 1). The vertebrate groups can have up to ten age classes that can contain multiple ages. The model tracks number per age within an age class and the weight of each age class which is divided into reserved and structural weight and is in mg N. Cephalopods and shrimp have two age classes, juvenile and adults. All other groups have no age classes and are modelled as biomass pools.

The initial conditions of most of the vertebrate groups, i.e. their estimated total biomass and weight per individual were acquired from data sampled by the Marine Research Institution (MRI) or from reports from the Marine Research Institute (2015).

Consumption and diet

The consumption rate of each group is modelled with the Holling type II function and the diet composition for each predator is adjusted with setting the availability of each prey. The spatial distribution of the predator and prey need to overlap in order for the predator to feed on the prey. A gape limitation is set for the predator to allow possible prey to become too big for the predator. The model allows for ontogenetic shifts in the diet, i.e. the juveniles can have a different diet than adults. Data from MRI on stomach content was used as a guideline when setting up the diet matrix. The food web can be seen in Figure 2.

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Table 1. Description of the functional groups.

Code	Vertebrates	Code	Invertebrates and other groups
FCD	Cod (*Gadus morhua*)	CEP	Cephalopod
FHA	Haddock (*Melanogrammus aeglefinus*)	PWN	Shrimp
FSA	Saithe (*Pollachius virens*)	ZS	Microzooplankton
FRF	Redfish (*Sebastes* sp)	ZM	Mesozooplankton
FGH	Greenland Halibut (*Reinhardtius hippoglossoides*)	ZL	Macrozooplankton
FFF	Flatfish	ZG	Gelatinous zooplankton
FHE	Herring (*Clupea harengus*)	LOB	Norway Lobster
FCA	Capelin (*Mallotus villosus*)	BML	Other Megazoobenthos
FMI	Blue whiting (*Micromesistius poutassou*)	SCA	Iceland Scallop
FOC	Other Codfish	QUA	Ocean Quahog
FDC	Other Demersal Commerical	CUC	Cucumbers
FDF	Other Demersal Fish	BD	Deposit Feeder
FSD	Sandeel Fish	BFF	Other Benthic Filter Feeders
FDL	Long Lived Demersal	BG	Benthic Grazer
FMP	Large Pelagic Fish	BC	Benthic Carnivore
FBP	Small Pelagic Fish	BO	Meiobenthos
SSR	Skates	PL	Diatom
SSD	Small Sharks	PS	Pico-phytoplankton
SSH	Large Sharks	MA	Macroalgae
SB	Seabird	MB	Microphytobenthos
PIN	Pinniped	SG	Seagrass
WMW	Minke Whale (*Balaenoptera acutorostrata*)	DF	Dinoflagellates
WHB	Whale Baleen	PB	Pelagic Bacteria
WHT	Whale Tooth	BB	Sediment Bacteria
WTO	Whale Tooth Other	DL	Labile detritus
		DR	Refractory detritus
		DC	Carrion

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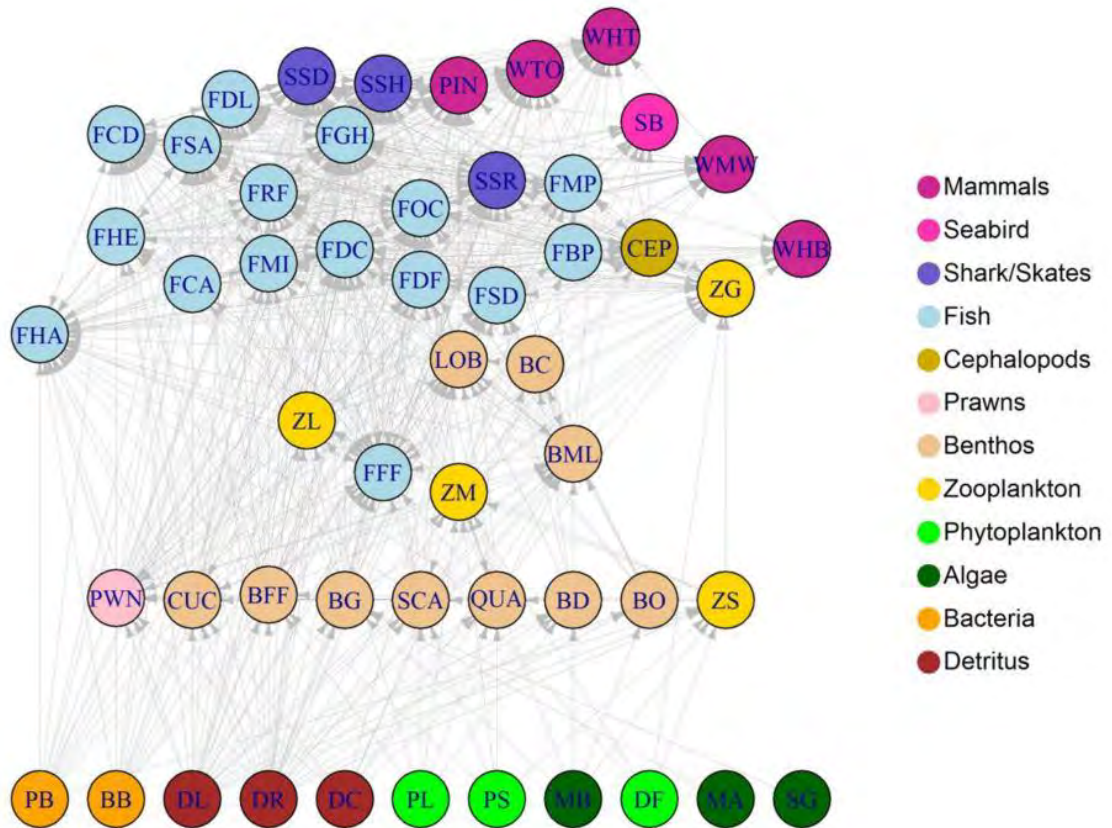development and demonstration under
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Figure 2. Food web from the Atlantis model for vertebrates of age class 4. See Table 1 for group code.

Recruitment

Recruitment of the fish groups was modelled with the Beverton-Holt function that describes the relationship between the spawning stock biomass and number of recruits. The recruitment of the mammals and the seabird groups was modelled as a constant per adult. It is possible to force recruitment spikes in Atlantis and this was done for the haddock as it controls the fluctuations in stock size.

Spatial distribution and migration

The functional groups can have different spatial distribution which can be different by season. The distribution was set as fixed for four different seasons and the model allows the distribution to be different between juveniles and adults. The spatial distribution of cod and capelin can be seen in Figure 3 to Figure 6. Groups can also migrate in and out of the model area. There are four migratory groups in the model: blue whiting, seabirds, minke whale and baleen whales.

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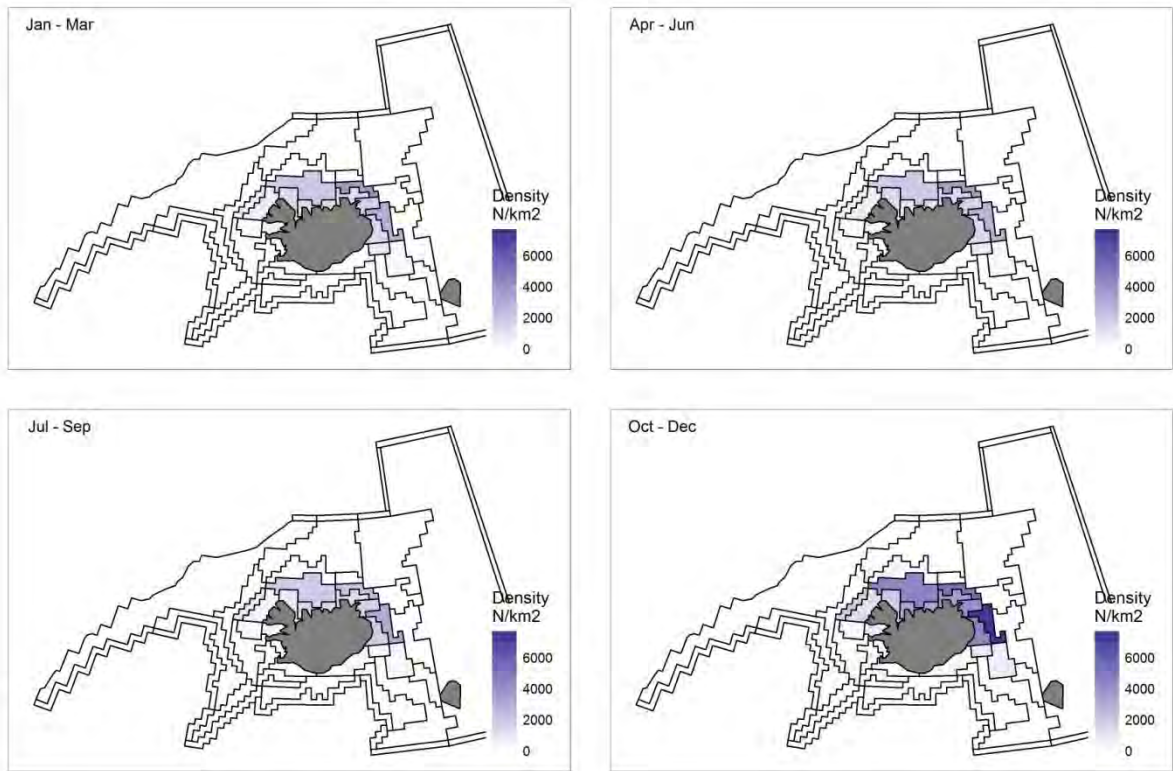grant agreement no. 613571

Figure 3. Spatial distribution of juvenile cod in the 1st age class for four seasons.
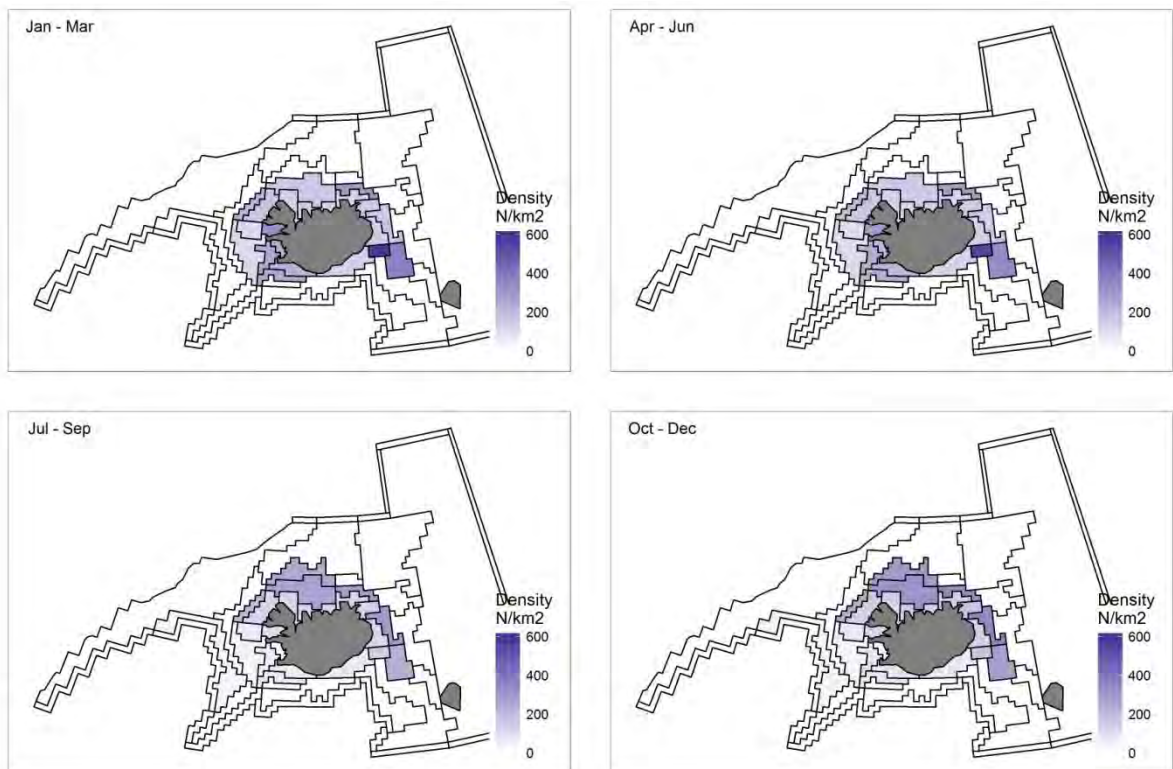

Figure 4. Spatial distribution of adult cod in 5th age class for four seasons.
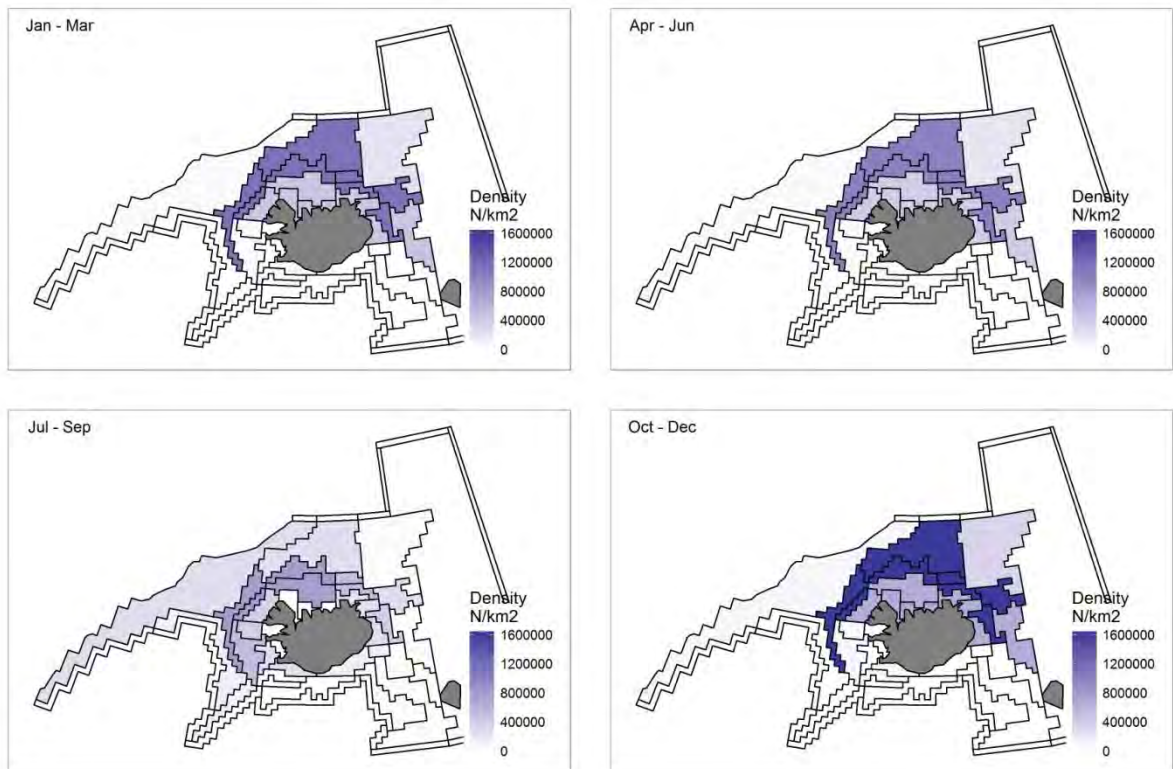
Figure 5. Spatial distribution of juvenile capelin in 1st age class for four seasons.
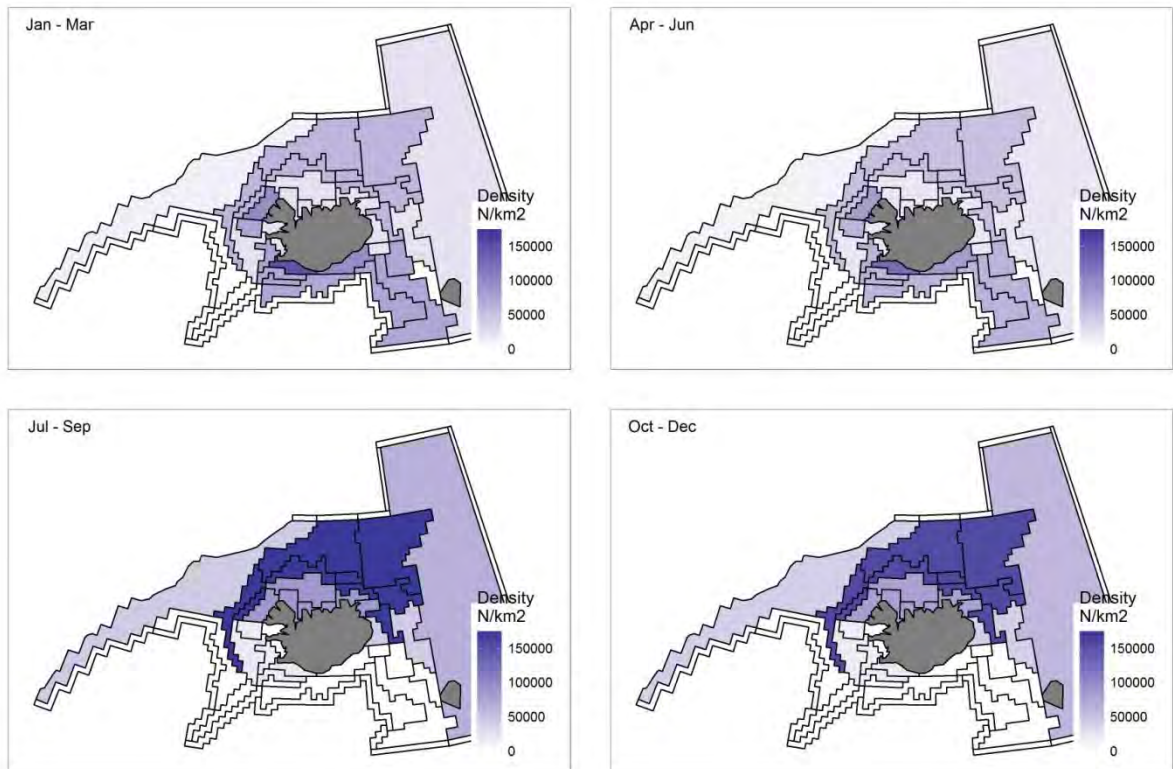

Figure 6. Spatial distribution of adult capelin in 3rd age class for four seasons.

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The fisheries model
The most important commercial species are harvested in the model. Each group is harvested by one fishing gear that has certain selectivity. The selectivity was chosen to be a logistic curve and size distribution of catch and survey data from the MRI was used to parameterise the selectivity curves. The harvest mortality is then multiplied with the selectivity curve which is based on length. The harvest mortality is allowed to change between years but the selectivity is the same for the whole period.

4.2.2. Indicators

Good environmental status indicators
GES indicators were calculated from the simulated model output. These indicators were: Indicator of stable biomass, trend of biomass and landings, harvest rate (HR), Shannon diversity index (SDI), Pelagic to demersal fish ratio (p/d), mean maximum length (MML), mean trophic level (MTL), marine trophic index (MTI), large fish index (LFI) and index of significant landings (SL). The calculation of these indicators will be described in this section.

Stable biomass
The biomass was considered stable if it did not change by more than 10% between years in the last five years of the simulation. This indicator was calculated for the eight most important commercial groups: cod, haddock, saithe, Greenland halibut, redfish, herring, capelin and blue whiting.

Biomass and landings trends
The trend of the biomass and landings was calculated for the last five years of the simulation and a regression line fitted. It was either determined downwards or upwards trend but significance was not tested. This was done for the same groups as for the stable biomass indicator.

Harvest rate
HR was calculated for the most important commercial groups: cod, haddock, saithe, Greenland halibut, redfish, herring, capelin and blue whiting. It was calculated as the total catch over total biomass (C/B) for each year of the simulation.

Shannon diversity index
The SDI of landings was calculated as follows:

$$SDI = \sum_{g=1}^{G} P_g \log_2 P_g$$

where P_g is the proportion of group g in total landings. SDI was calculated for each year of the simulation and included the eight most important commercial groups mentioned above.

Pelagic to demersal fish ratio
Pelagic to demersal ration was the total biomass of pelagic fish groups divided with the total biomass of the demersal fish groups. Pelagic fish groups were considered to be: herring, capelin,

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blue whiting, small pelagic and large pelagic. Demersal fish was: cod, haddock, saithe, Greenland halibut, redfish, flatfish, other codfish, demersal commercial, other demersal, long lived demersal, sandeel, skates, small sharks and large sharks.

Mean maximum length

The MML was calculated for the groups that are mostly influenced by fishing. These are: cod, haddock, saithe, Greenland halibut, redfish, herring, capelin and blue whiting. The Atlantis model tracks weight of each age class within a functional group. The weight is converted to length using the length weight relationship ($W = aL^b$). The maximum length is found for each group each year and weighted average calculated based on the total biomass of each group.

Mean trophic level

The Atlantis output provides the diet composition at each time point. From the diet composition, which is proportion of each prey by weight of each consumer group, the trophic levels are calculated using the FlowBasedTrophicLevel function in the Cheddar package (Hudson et al., 2015) in R (R Core Team, 2016). Atlantis allows for ontogenetic shifts and therefore the diet composition can be different between age classes. This make calculation of trophic levels more complicated and to simplify the calculations one age class is chosen to represent the group. The vertebrate groups usually have ten age classes and the diet composition of age class four was used to calculate the trophic level. The weighted average of the trophic level was calculated based on the total biomass of each group. The MTL for time t is calculated as follows:

$$MTL_t = \frac{\sum TL_{it} * B_{it}}{\sum B_{it}}$$

where TL is the trophic level at time t for group i and B is the total biomass of group i at time t.

Marine trophic index

The trophic level was calculated as described for MTL and the mean trophic level is weighted by the landings. The MTI for time t is calculated as follows:

$$MTI_t = \frac{\sum TL_{it} * Y_{it}}{\sum Y_{it}}$$

where TL is the trophic level of group i at time t and Y is the landings of group i at time t.

Large fish index

Large fish was considered to be larger than 60 cm. The biomass of fish which was larger than 60 cm was calculated and divided with total fish biomass as described below.

$$LFI = \frac{\sum B_{i,le>60}}{\sum B_i}$$

Significant landings

Number of groups with SL was number of groups which had HR higher than 5%.

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Economic indicators

Demersal fisheries

Economic indicators were calculated for five demersal groups using a method described in Hoskuldsson et al. (2015). Using this method the revenue, cost and profit were calculated. It was assumed that the cost of oil and gear (C_1) is 55 ISK per kg of yield of demersal fish in the year 2000 and has inverse relationship with biomass (Eq. 1). The cost of owning and maintaining a ship (C_2) was assumed to be 53 ISK per kg of yield in 2012 and having an inverse relationship with biomass (Eq. 2).

$$C_1 = 55 * \frac{B_{c,2000}}{B_c} \qquad (1)$$

$$C_2 = 53 * \left(\frac{B_{c,2012}}{B_c}\right)^{0.8} \qquad (2)$$

The price of fish from 2012 was taken from Verðlagsstofa skiptaverðs (2016). The prices depend on the weight of the groups: cod, haddock, saithe and redfish but the average price was only given for Greenland Halibut. A fixed price is given if the fish is under the minimal legal size (Figure 7).
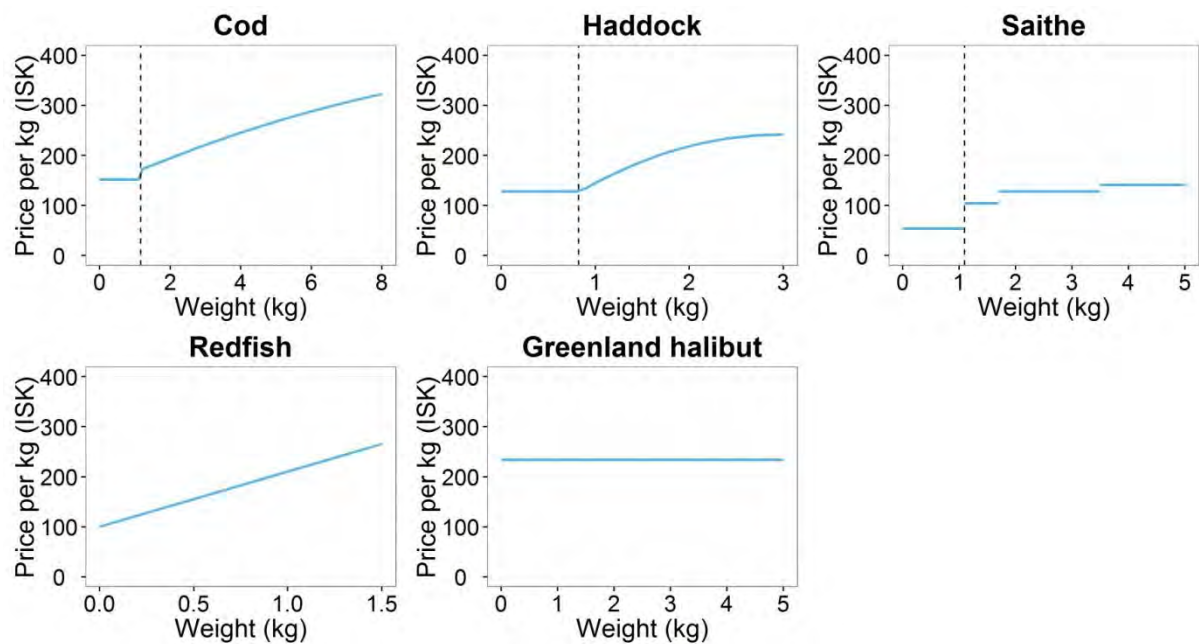
The revenue (R) from each group is given as:

$$R = \sum_w Y_w * p_w$$

where Y is the total catches of a certain weight group and p is the price for that group. The cost (C) depends on the catches and are as follows:

$$C = Y(C_1 + C_2)$$

The profit (P) then becomes:

$$P = R - C$$

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Figure 7. The price per kg of cod, haddock, saithe, redfish and Greenland halibut. The dotted line shows minimum legal size.

Pelagic fisheries

The most important pelagic species with commercial value are herring, capelin, and blue whiting. These species have a schooling behaviour and costs associated with these fisheries have different characteristics than in the demersal fisheries (Sandberg, 2006). In bottom trawl fisheries 63% of the oil cost is because of trawling but herring fisheries only use 5.2% of the total oil for actual fishing (Björnson, 2004). Therefore a different method for calculating economic indicators is used for the pelagic groups.

In some herring fisheries in Norway the stock size did not have a significant effect on the cost of fishing (Sandberg, 2006). It will therefore be assumed that the cost is constant per ton of harvested pelagic fish. This constant cost is however assumed to be different between the fisheries. It has been estimated that the oil use associated with the fisheries is 20, 82 and 90 l per ton for capelin, herring and blue whiting, respectively (Björnsson, 2004). There is no estimate for mackerel but it will be assumed that the cost is the same as for the herring as it is harvested closer to land than the blue whiting.

In 2012 oil cost for pelagic trawler (freezer-trawler excluded) in Iceland was 3,839 million ISK (Statistics Iceland, 2016). That same year 126, 456, 40, 67 thousand tonnes were harvested by these trawlers of herring, capelin, blue whiting and mackerel, respectively. The cost of oil can then be divided between the different fisheries and oil cost per ton harvested calculated (Table 2). Total cost of fishing gear was 1,135 million ISK and total cost of maintenance was 1,487 million ISK in 2012 (Statistics Iceland, 2016). The cost of gear and maintenance is assumed to be the same for the four fisheries (see Table 2). Mackerel is not included in the model but this species invaded Icelandic waters in 2007.

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Table 2. Price per kg and cost per ton by pelagic fisheries in 2012.

Species	Price (ISK/kg)	Oil cost per ton (ISK)	Fishing gear cost per ton (ISK)	Maintenance cost per ton (ISK)	Total cost per ton (ISK)
Herring	49	11,010	1,647	2156	14,813
Capelin	25	2,685	1,647	2156	6,488
Blue whiting	26	12,084	1,647	2156	15,887
Mackerel	52	11,010	1,647	2156	14,813

The revenue (R) is then as follows:

$$R = p * Y$$

The cost is a function of the catch which is different for the fisheries:

$$C_{herring} = 14{,}813 * Y$$

$$C_{capelin} = 6{,}488 * Y$$

$$C_{blueWhiting} = 15{,}887 * Y$$

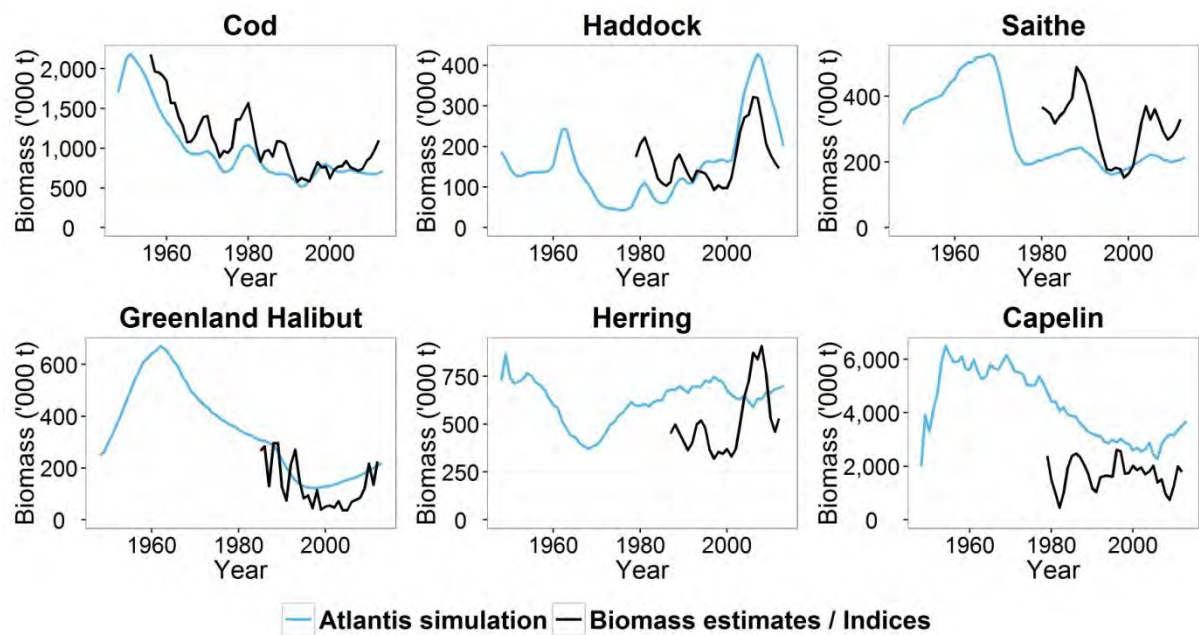The profit for species *s* then becomes:

$$P_s = R_s - C_s$$

Social indicators

Salaries to fishermen are used as a social indicator. Fishermen salaries in Iceland are connected to the total revenue. According to collective agreement (Sjómannasamband Íslands, 2008) the crew share of the revenue depends on the size of the vessel, fisheries and crew size. Each vessel usually participates in either demersal or pelagic fisheries but does not specialize in one species. Therefore the indicators are not given for each species. In 2012 a crew working on trawler in demersal fisheries got 25% of the total revenue but 27% in pelagic fisheries (Statistics Iceland, 2016) and that will be used to calculate fishermen salaries from the Atlantis model.

4.3. Results and discussion

4.3.1. Simulated biomass and catches

Simulated biomass from the model was compared to biomass estimates from MRI (Marine Research Institute, 2015). These estimates are for the fishable stock or the spawning stock and have been scaled in Figure 8 to take into account the total population. The stock size of cod, haddock, saithe, herring and capelin are estimated annually. Estimates are also available for other species such as golden redfish, plaice, ling and Atlantic wolfish but these species are all included in functional groups which also have species with no biomass estimates and comparisons are therefore not attempted. For other species an index is measured. These include Greenland halibut, beaked redfish, spotted wolfish and more species (see Marine Research Institute, 2015). Only the Greenland halibut is modelled as a single species and the simulated biomass is compared to a scaled index (Figure 8).

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Figure 8. Simulated biomass of Cod, Haddock, Saithe, Greenland Halibut, Herring and Capelin from 1948 to 2012 compared to estimated biomass, except Greenland Halibut is compared to a scaled indices.

The model is able to simulate the trends and magnitude of biomass for cod, haddock and Greenland halibut. The simulated biomass is not far from the estimates for saithe but it has been very difficult to estimate the size of the saithe stock because of it unpredictable behaviour. The model does not get the trends of the herring stock. The trend may be influenced by the recruitment as is the case for the haddock where recruitment spikes have been forced in the model. Herring does not have a forced recruitment but this may be needed to simulate the trends for this group. Capelin is another group where the Beverton-Holt function may not be the best option to model the recruitment.

The harbour seal, grey seal, minke whale and fin whales are counted regularly (Marine Research Institute, 2015). The biomass estimates are calculated by multiplying the average weight (Gunnarsson er al., 1998; Sigurjónsson & Víkingsson, 1997) with the abundance. The simulated biomass of the seabird and mammals group can be seen in Figure 9. The pinnipeds, minke whales and baleen whales are compared to biomass estimates. The simulated biomass is in the same magnitude as the estimates but does not catch the trends. Note that the estimate of baleen whales is only based on fin whale but other baleen whales are considered in the group so the estimates should be lower than the simulated biomass. Harvesting of pinnipeds is not included into the model but reported catches have been as high as 6000 seals in a year.
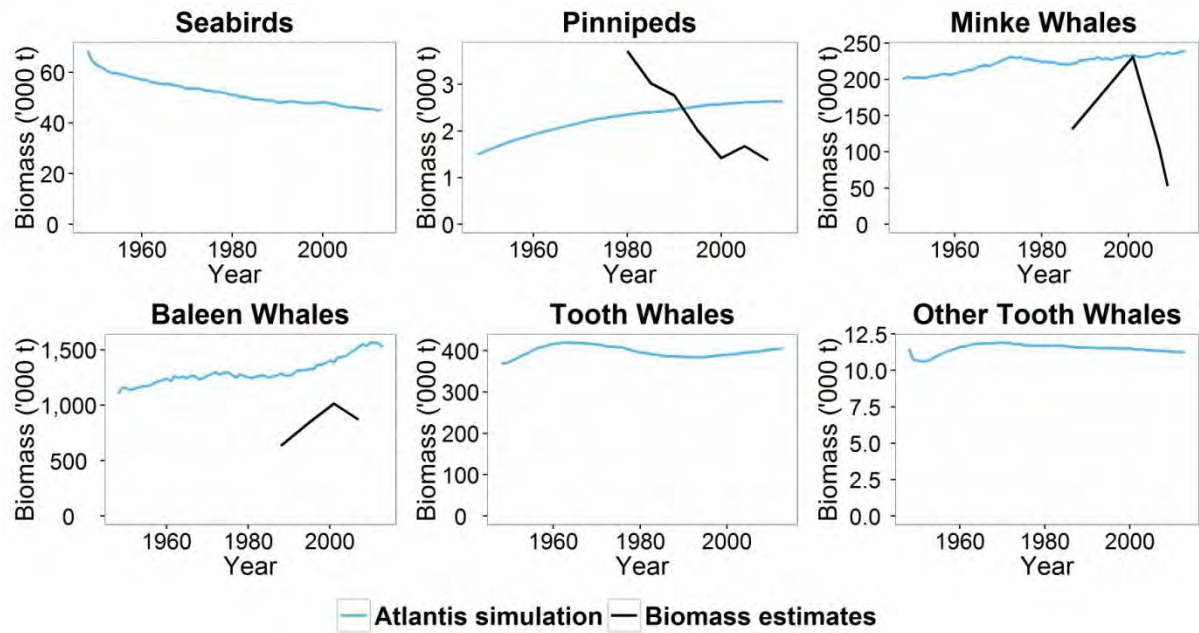
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Figure 9. Simulated biomass of Seabirds and mammal groups from 1948 to 2012.

Simulated biomass of other vertebrates groups was not compared to biomass estimates as they were not available for all the species in the groups (Figure 10). The biomass of redfish and blue whiting drops at the time harvesting of these groups begins. The top predators, e.g. shark increase in biomass over the simulated period. The mortality of these groups is controlled by linear and density dependent mortality parameters as other groups do not feed on the top predators. These mortality parameters may have to be further tuned to avoid the increase in stock size. The simulated biomass of the small pelagic fish increased around 1980. This can be explained by decrease in stock size of their predators such as cod and redfish.
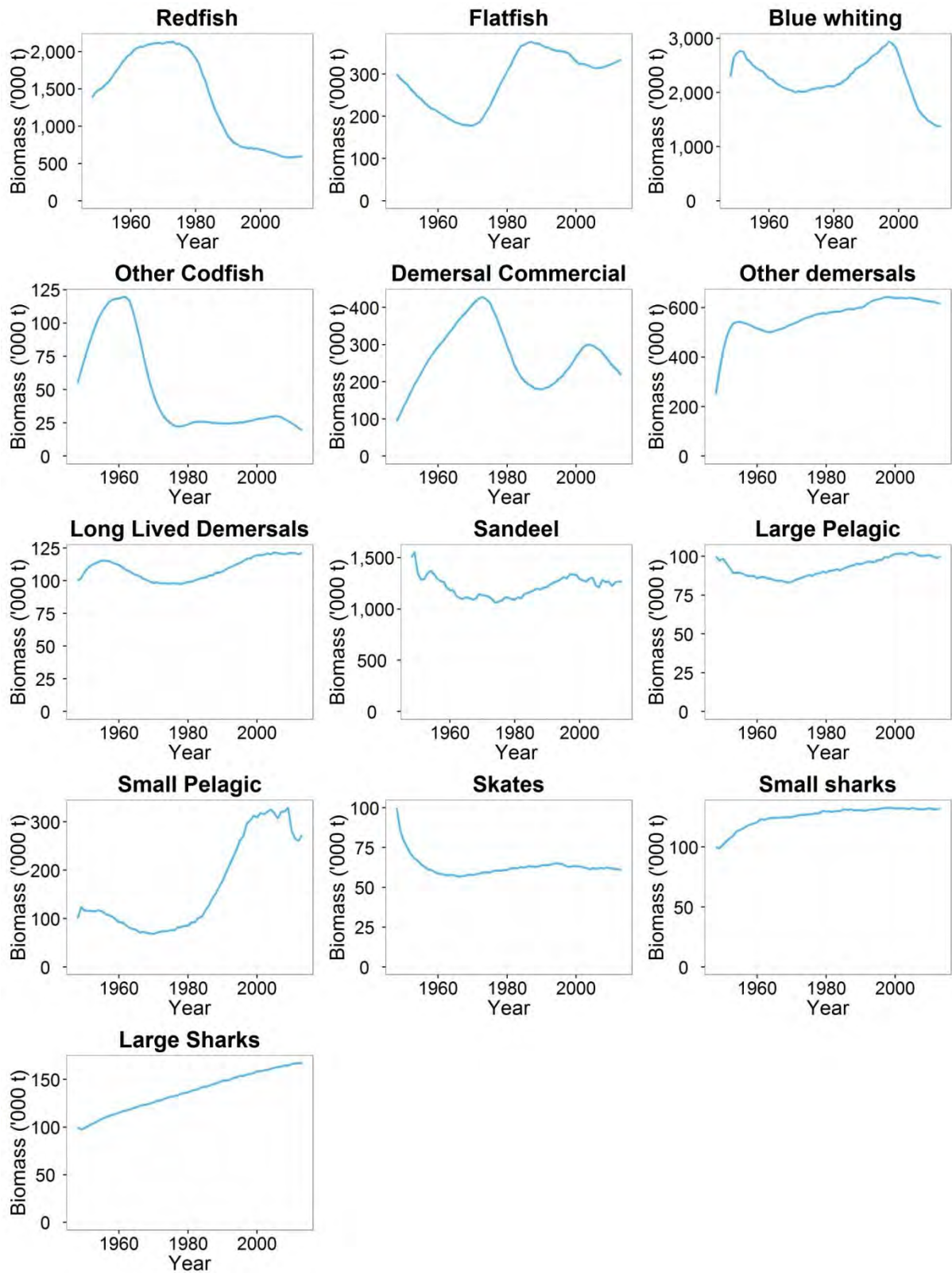
Figure 10. Simulated biomass of fish groups from 1948 to 2012.

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The simulated catches from the model is compared to landings data (Figure 11). The model is able to catch the trends and magnitude for most of the groups. Still, some further calibration is needed. Parameters in the biological processes that have influence on stock size have consequently also influence on the catches. The harvest rate and selectivity curve of the fisheries also have effect on both the stock size and catches and these can be improved further.
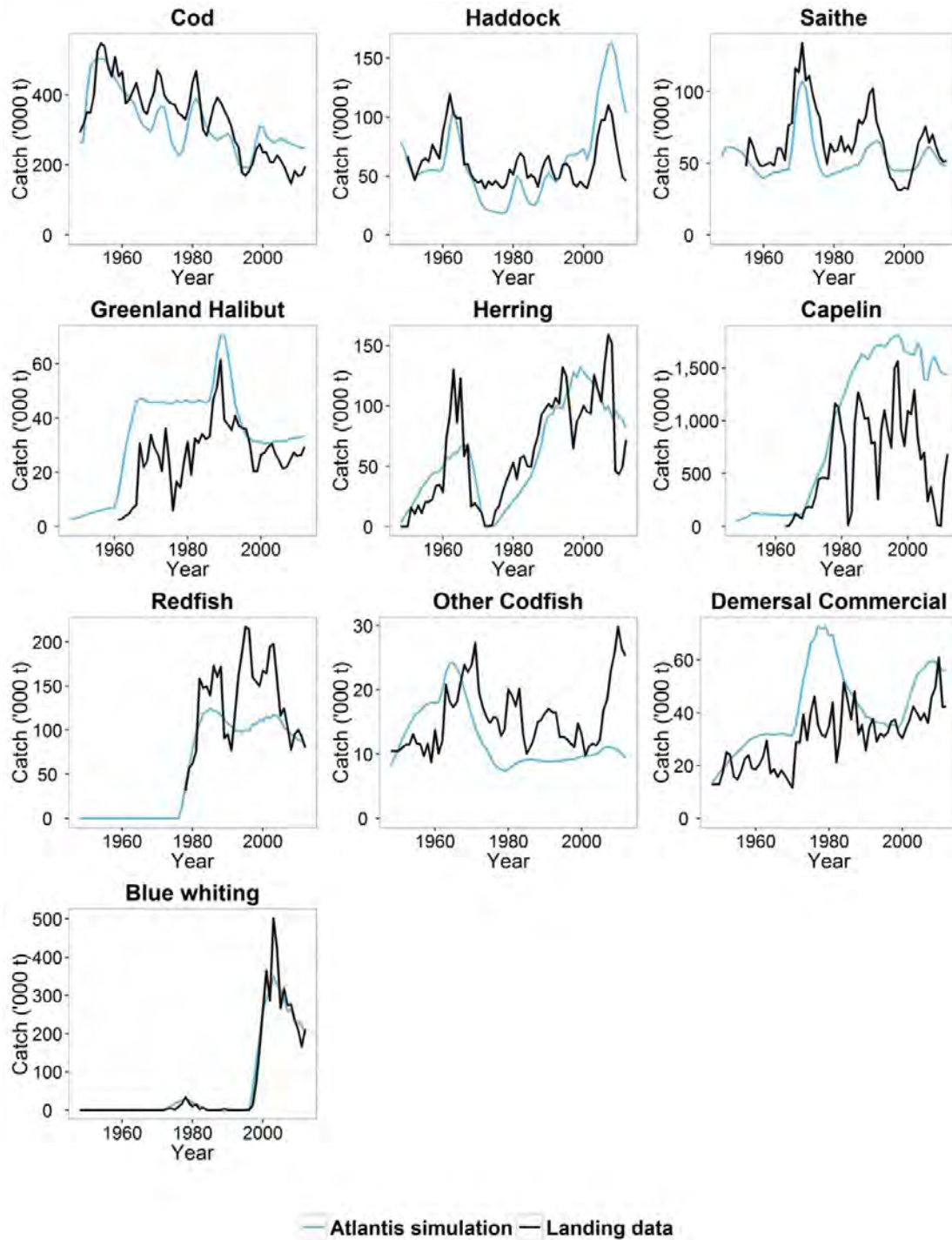

Figure 11. Simulated catch of commercial groups from 1948 to 2012 compared to landing data.

4.3.2. Indicators

Good environmental status indicators

The GES indicators where calculated from the simulated output from the model. The most important commercial groups had stable biomass with upward biomass trends and decreasing landings (Table 3) which indicates that the stock size is increasing slowly but steadily and the landings should therefore start to increase. The exception from this was the haddock which had unstable biomass and decreasing biomass trend. This decrease can be explained by recruitment fluctuations but not increasing fishing pressure. The landings of the Greenland halibut have been increasing but this should not be because of increasing fishing pressure but because the biomass has been increasing.

Table 3. GES indicators of stable biomass and trends in biomass and landings for eight commercial groups.

Group	Stable biomass	Biomass trend	Landings trend
Cod	Yes	Up	Down
Haddock	No	Down	Down
Saithe	Yes	Up	Down
Redfish	Yes	Up	Down
Greenland halibut	Yes	Up	Up
Herring	Yes	Up	Down
Capelin	Yes	Up	Down
Blue whiting	Yes	Down	Down
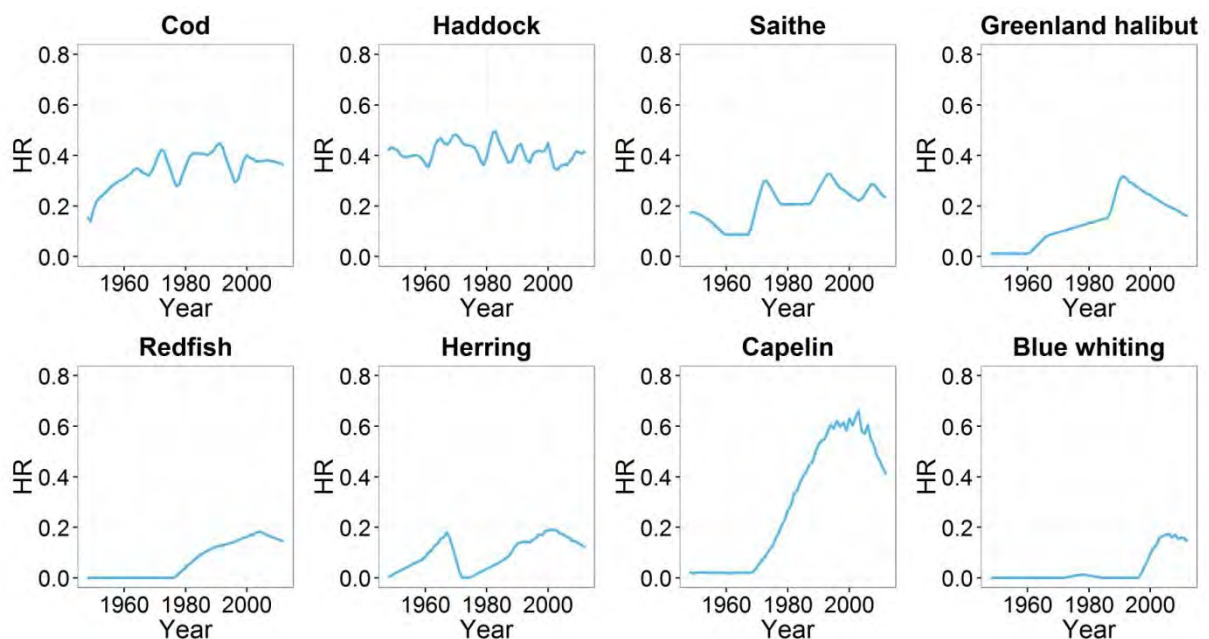
Figure 12. Harvest rate from the model output for cod, haddock, saithe, Greenland halibut, redfish, herring, capelin and blue whiting.

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The HR is shown in Figure 12 and the groups have a HR from 15% to 60%. The short lived species such as capelin can tolerate much higher HR than the redfish which is a long lived species. The result of this simulation did however show a decrease in the capelin stock when harvesting of that species began. Other GES indicators are shown in Figure 13. The model is highly suitable to calculate these indicators as the output contains numbers and size by age classes of the vertebrate groups. It also gives annual diet composition which can be used to calculate the trophic level. It should be noted that no burnout time was used in the simulation. Therefore there can be imbalance at the beginning of the simulation run as can be seen for the MTL indicator. In the first year of the model run the biomass of cephalopods increases threefold and then drops down to its initial value next year. The cephalopods with a trophic level of 3.8 have very high biomass and therefore pull the MTL upwards in the first year.
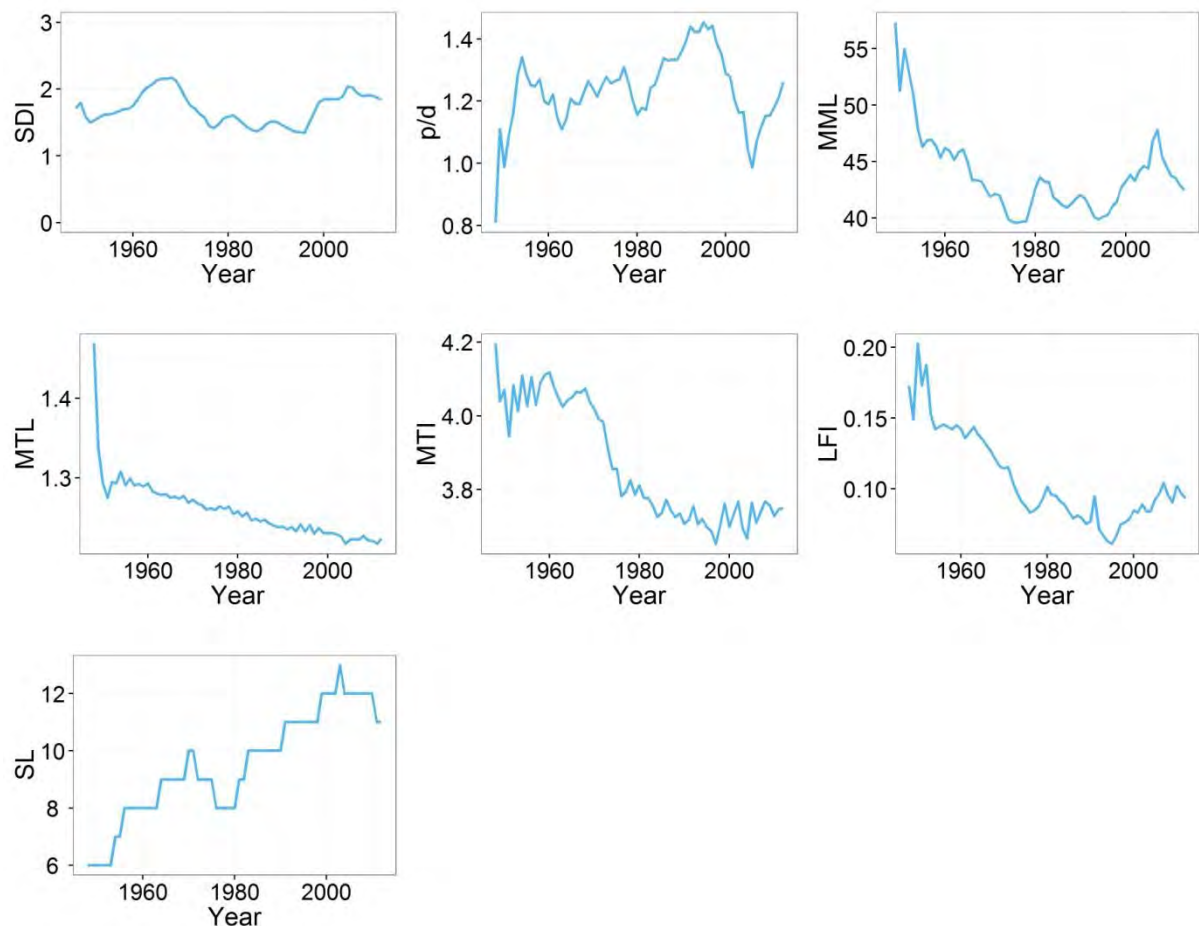

Figure 13. Good environmental status indicators: Shannon diversity index (SDI), Pelagic to demersal ratio (p/d), mean maximum length (MML), mean trophic level (MTL), marine trophic index (MTI), large fish index (LFI) and number of significant landings (SL).

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Economic indicators

The economic indicators, revenue, cost and profit are shown for demersal species in Figure 14 and for pelagic species in Figure 15. All the groups showed positive profit over the simulated period except haddock had negative profit when the stock size was lowest. The simulated biomass and the catches are however lower than estimates (Figure 8 and Figure 11) so the profit is underestimated. It also should be noted that the demersal species are usually not caught separately but catches of cod, haddock and saithe frequently goes together. The cod is the most important commercial species with revenue and profit of 75 and 40 billion ISK, respectively, at the end of the simulation period but much higher when the stock was at its peak around 1953. The redfish had a profit of 20 billion ISK at the begging of the redfish fisheries but the profit dropped as the cost of fishing increased with decreasing stock size.

The calculation of the cost for the pelagic fisheries is only a function of the harvest. This may be too simple, e.g. if time searching for fish schools takes longer time when stock size is low. Sandberg et al. (2006) did however not find a significant relationship between stock size and cost for some pelagic fleets. These results show that the capelin fisheries are ten times more profitable than the herring and blue whiting fisheries. The oil cost for the separate pelagic fleets is based on limited data (see Björnsson, 2004) and the cost for the herring fisheries may be overestimated and the cost of capelin fisheries hence underestimated. The cost and profit of total pelagic fisheries should however show realistic results.

The revenue of the total fisheries has fluctuated over the period but has not showed a consistent trend (Figure 16). The cost has on the other hand showed an increasing trend resulting in lower profit at the end of the simulation period.
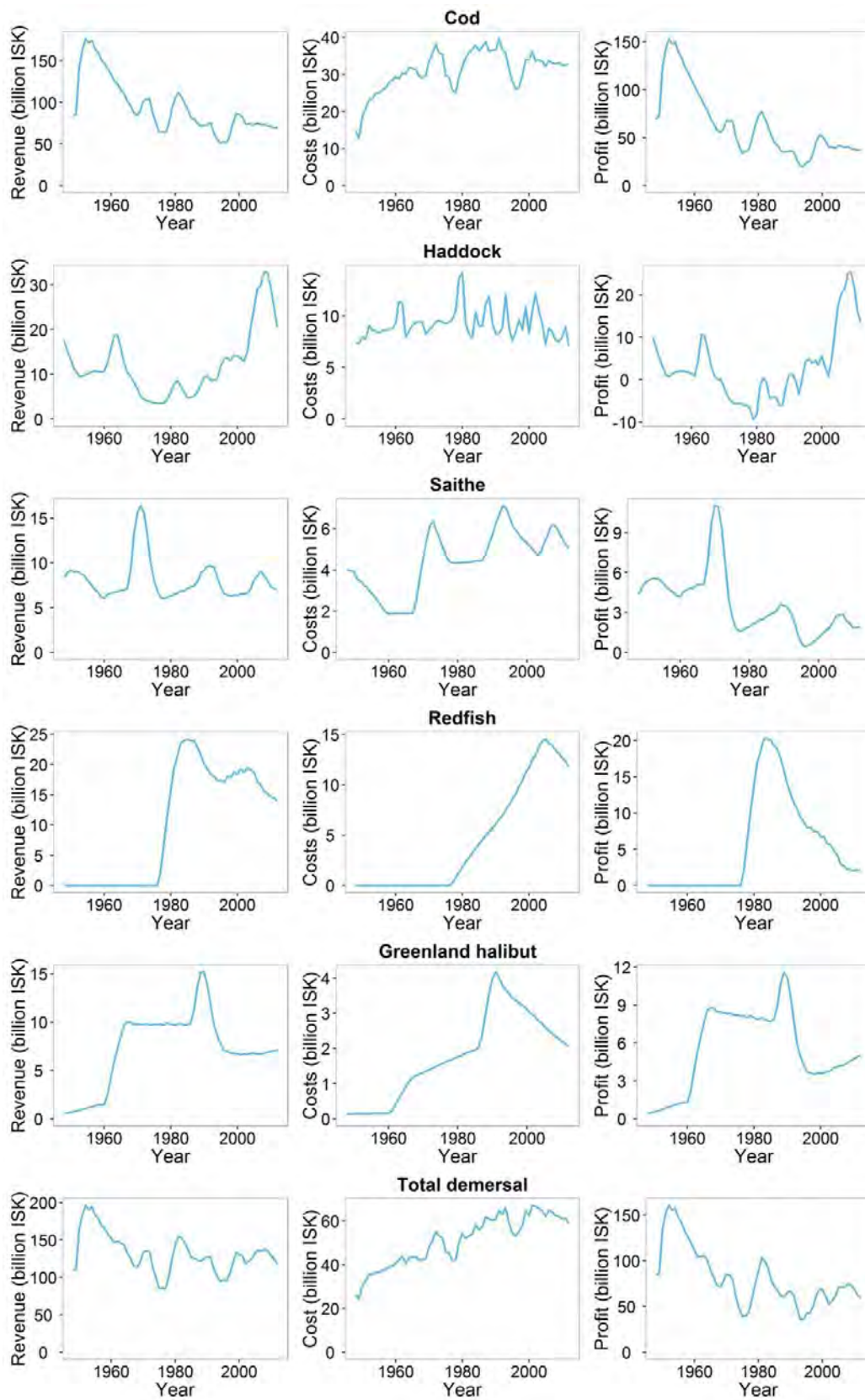
Figure 14. Revenue, costs and profit from demersal fisheries (cod, haddock, saithe, redfish and Greenland halibut).
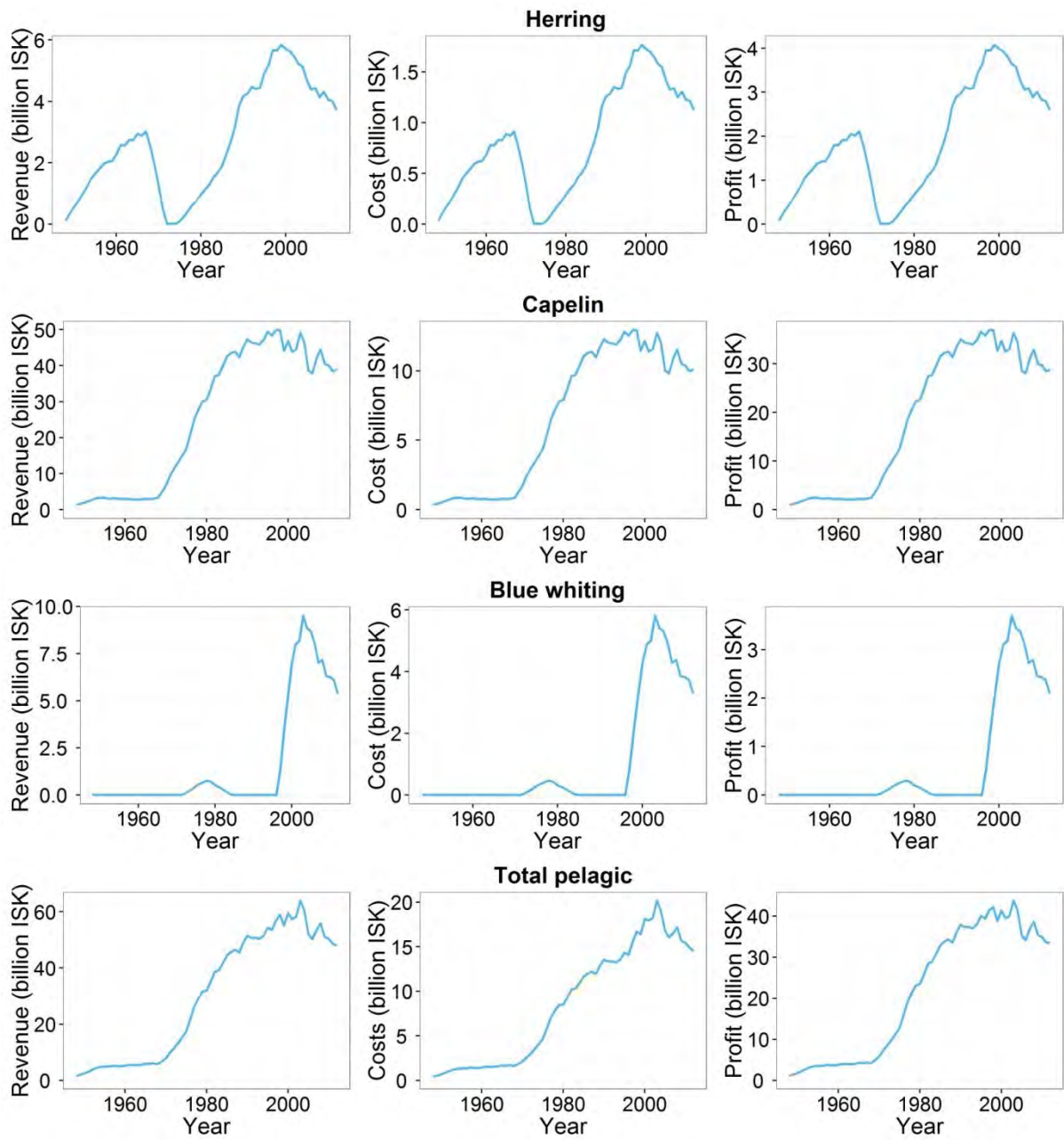
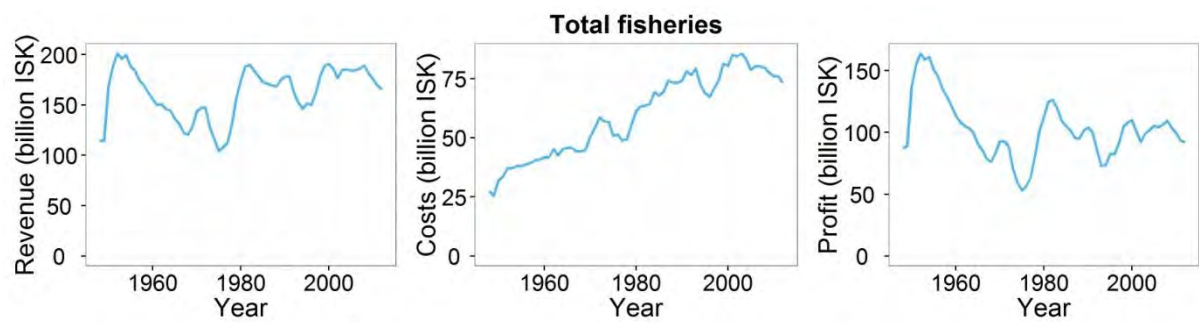**Figure 15. Revenue, cost and profit from pelagic fisheries: herring, capelin, blue whiting.**

Figure 16. Total revenue, cost and profit for demersal and pelagic fisheries combined.

Social indicators

The social indicator used in this case study is the total salaries the crew receives. These salaries are a certain proportion of the total revenue as described in section Indicators4.2.2. and may not necessarily reflect the number of jobs. This however shows how much goes into the economy as fisherman salaries.
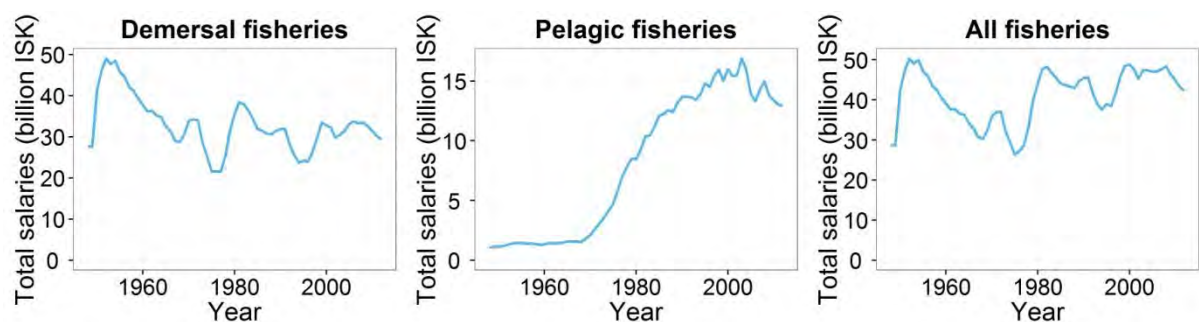

Figure 17. Total salaries for demersal and pelagic fisheries.

4.4. Conclusion

An Atlantis model has been built for Icelandic waters that is able to simulate realistic biomass and landing trends for the most important commercial groups. The indicators described in this study can help to identify the ecological and socio-economic effects a change in fishing pressure might have. Building a model like the Atlantis model is a continuous process as the model can always be improved. The current model would benefit from calibration of parameters but it still shows realistic output and can be used for scenario testing. A whole-of-an- ecosystem model like the Atlantis model has a great potential to explore effects of fishing on the whole ecosystem and hence support the ecosystem approach to fisheries management.

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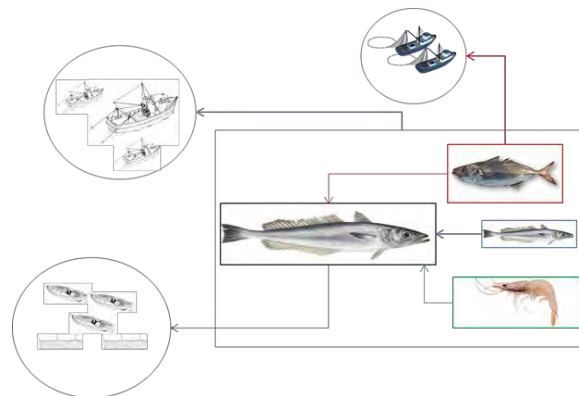# 5. STRAIT OF SICILY SEA CASE STUDY: GADGET

5.1. Introduction

The Strait of Sicily (SoS) case study (CS) focuses on bottom trawl fisheries exploiting deep water rose shrimp (DPS: *Parapenaeus longirostris*) and hake (HKE: *Merluccius merluccius*) in GSAs 12-16 as described in D 5.3. GADGET (Beagley and Howell, 2004) was used as alternative model focusing on trophic flows between hake, horse mackerel (HOM: *Trachurus trachurus*) and deep water rose shrimp to improve the understanding of the dynamics of these stocks under different management scenarios. These were linked to four main objectives identified in co-creation during the case study meetings: i) rebuilding overexploited stocks; ii) long-term continuity of the fishing activities; iii) same rules for all; iv) good environmental status. In particular the GADGET model for the SoS area was designed to understand the impact of a reduction in fishing mortality/annual catches, as effect of reduction in fishing effort, in a multifleet context. The model includes time series of catches from Italian and Tunisian fishing fleets, trawl survey data (MEDITS) and it is based on knowledge of the trophic interaction between species.

The aim of this section is to compare simulated time-series of total biomass, fishing mortality and catches of the two main stocks (DPS and HKE) from the model with estimated biomass from stock assessments and official catch data. GES and economic will be also calculated for the simulated period.

5.2. Methods

5.2.1. Multispecies GADGET model

Gadget in SoS is designed to model interactions between 2 fish populations (HKE and HOM), 1 shrimp population (DPS) and 5 main fleets: Italian and Maltese trawlers, Tunisian trawlers, Italian and Maltese small-scale vessels, Tunisian small-scale vessels, Italian and Tunisian purse seiners and mid water pair-trawlers. Hake cannibalism is also included (Fig. 6.2.1). Fleets subtract biomass in different ways from the three populations and display differences in the exploitation pattern. Hake is the predator of HOM, DPS and itself. Bottom trawlers target DPS and HKE having HOM and HKE juveniles as by-catch. Mid-water trawlers and purse seiners have HOM as by catch. Artisanal vessels catch HKE. Details on model specifications and input data can be found in D 5.3.

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Figure 6.2.1. Conceptual structure of the multispecies GADGET model for hake (HKE), horse mackerel (HOM) and deep water rose shrimp (DPS), with HKE predator of HOM, DPS and HKE (cannibalism). Fleets exploiting the three stocks: a) bottom trawlers; b) purse seiners and mid-water trawlers, c) artisanal vessels.

5.2.2.   Settings and assumptions

HKE and HOM populations are defined by 2 cm length groups, whilst DPS by 3 mm carapace length groups. The year is divided into four quarters. HKE age range is 0 to 7 years, with the oldest age treated as a plus group. Recruitment happens in the second, third and fourth quarter. The length at recruitment is estimated and mean growth is assumed to follow the von Bertalanffy growth function (VBGF) with L_{inf}=100 and K estimated by the model. DPS age ranges in between 0 and 3, this latter used as plus group. Recruitment takes place all year round. HOM age range is 0 − 6+. Parameters of VBGF are L_{inf}=44 (fixed) and K=0.23 estimated. Models parameters are listed in ANNEX 6.3. Natural mortality was assumed as a vector using the PRODBIOM approach (Abella et al., 1997) by an ad hoc code implemented in R.

Fleet selectivity curves

Commercial data include annual catches and size frequency distribution of Italian, Maltese and Tunisian trawlers (HKE and DPS) for the period 2002-2014. Artisanal fleets landings and size distributions are more scattered in time and included in HKE and HOM models. Survey data (MEDITS bottom trawl survey) cover the period 2002-2014 (Table 6.3.2).

Native GADGET functions were firstly used to estimate the fleets' selectivity (or suitability) model parameters for hake. However, considering that big hake has a reduced trawl catchability (see Bartolino et al., 2011) a new double logistic function was developed. It assumes a dome shape with a constant (at some level) right tail, to reproduce a catchability decreasing after a given size up to a constant level.

The new function is the following:

$a_l, a_r, l_{50}, r_{50} > 0, \quad l_{50} < r_{50}, \quad 0 \leq p \leq 1, \quad L > 0$ and

$$l_{const} = \begin{cases} L - r_{50} - x, & if \ L > r_{50} - x \\ 0, & otherwise \end{cases}$$

where $x = \log((1-p)/p)/a_r$, we define this new suitability function as

$$S(L; a_l, a_r, l_{50}, r_{50}, p) = \frac{1}{[1+exp(-a_l(L-l_{50}))] * [1+exp(a_r(L-r_{50}-l_{const}))]}$$

In the above formulation, parameters a_r and r_{50} play the same role in the right tail as the corresponding parameters a_l and l_{50} for the left side, while p indicates the proportion of fish captured after length $r_{50} + x$ (Fig. 6.2.2)
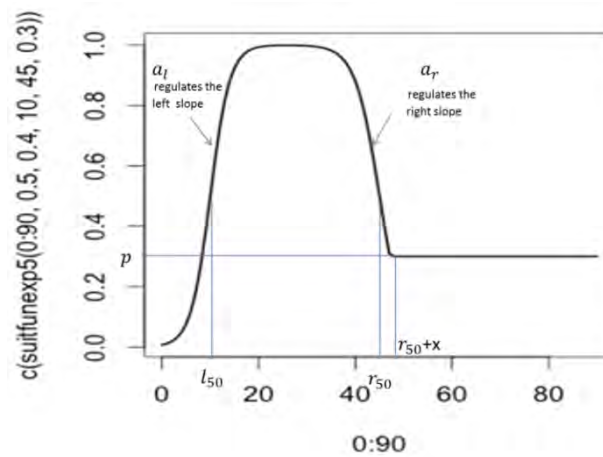
Figure. 6.2.2. The dome-shape and constant right tail selectivity function assumed for modelling the hake capture by bottom trawl fleets.

Sigmoid logistic selectivity functions were adopted for both DPS and HOM. The parameters of the selectivity curves estimated by GADGET single species models were fixed for multispecies GADGET parametrization.

Consumption

Prey consumption rate C is modelled in GADGET as dependent on the length of both the predator and the prey p, as well as the relative abundance of the prey (when compared to the total amount of food available). Values of C can affect predator growth depending on the growth function selected. The consumption equations are formulated in a flexible form as follows (Begley, 2005):

$$C_p(l,L) = \frac{N_L M_L \varphi_L F_p(l,L)}{\sum\limits_{p} F_p(l,L)}$$

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The parameter F_p (l,L), gives the amount of a given prey that is consumed by the predator, which is obtained by multiplying the biomass and energy content E_p of the prey by the suitability S, such that: (see below).

$$F_p(l,L) = \left(S_p(l,L) E_p N_l W_l \right)^{d_p}$$

M represents the maximum possible consumption for the predator and depends by temperature and length as follow:

$$M_L(T) = m_0 e^{\left(m_1 T - m_2 T^3 \right)} L^{m_3}$$

Where m_1, m_2 and m_3 are constants.

Finally φ is the "feeding level":

$$\varphi_L = \frac{\sum\limits_p F_p(l,L)}{H + \sum\limits_p F_p(l,L)}$$

where:

L is the length of the predator;

l is the length of the prey;

H is the half feeding level (i.e. the biomass of prey required to allow the predator to consume prey at half the maximum consumption level);

d is the preference of the predator for the prey;

N is the number of prey in the length cell l

W is the mean prey weight in the length cell

T is the temperature.

For hake in SoS the maximum consumption (M) as a function of predator length was calculated using a simplified bioenergetic model based on the approach developed by Temming and Hermann (2009) and already applied during the EU BECAUSE project.

The parameters of the von Bertalanffy growth function in weight (VBWF) were calculated using the following equation:

$$W_t = W_\infty \times \left(1 - e^{-\frac{3 \times D}{b} \times K \times \left(t - t_o \right)} \right)^{\frac{b}{D}}$$

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where b is the exponent of the length-weight relationship for sex combined (W=0.004TL^3.15). D is equal to b-a (a represents the length exponent of the physiologically limiting surface, $a = 2$ in the conventional VBGF).

The D value adopted for cod was 0.6. Consumption rate (F), F = dC/dt = consumption in g/day, was calculated using the following equation:

$$F = \frac{1}{K_3} \times 3 \times K \times W_\infty^{\frac{D}{b}} \times W^{\frac{a}{b}}$$

where:

K_3, the Ivlev coefficient of energy utilisation of third order, was fixed to 0.55;

K is 0.12 from the von Bertalanffy growth curve for the two sexes combined;

$W\infty$ is 7980 g

$a/b = m$, the allometric exponent of consumption, is equal to 1- b/D. Its value generally range between 0.67 and 0.9, 0.8 for cod and whiting (Temming and Herrmann, 2009). For hake in SoS we fixed a/b =0.8.

The relationship in Fig. 6.2.3 was used to calculate the parameters of the maximum consumption at length, assuming that this can be 1.5 higher than the average consumption at length estimated for the stock. Consumption at length was expressed as:

ML=$m_0 L^{m3}$ (see Lindstrom et al., 2009):

where m3 = 2.52 (i.e. the exponent m of 0.8 multiplied by the exponent b of the length-weight equation b=3.15) and m0=0.071 (grams).
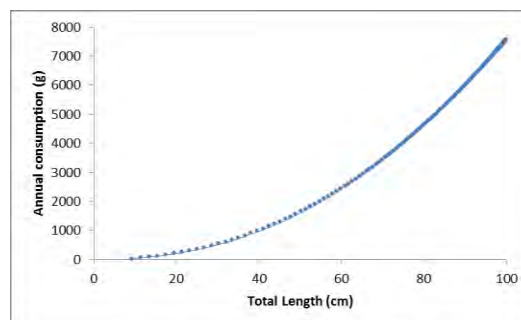

Fig. 6.2.3. Annual consumption curve for hake in the length range between 4 and 100 cm TL

The suitability function for consumption used for the SoS GADGET model was based on a modified version of Andersen and Ursin (Andersen and Ursin, 1977). The original Andersen and Ursin (AU) function assumes the consumption is dependent on the ratio of the predator length

to the prey length. In the AU formulation, parameter p_2 is a scalar which determines the maximum suitability for the particular prey, this may be lower than 1 for a non-preferred prey type. In the present model, we adjusted the p_1 values for HKE, HOM and DPS based on the diet composition in weight of hake 5 cm length classes in 2013-2014. However, whatever the predator length is, the maximum consumption level is assumed constant at a level proportional to p_2p_2. To account for a diffential prey preference of hake during growth, as observed from stomach contents data, and thus to allow the predator maximum suitability level to vary with prey length, we multiplied parameter p_2 to a function $\hat{y}(L)$, depending on the predator length:

$$S(l,L)=\begin{cases} p_0 + p_2\hat{y}(L)\exp\left[-\dfrac{\left(\log\frac{L}{l}-p_1\right)^2}{p_3}\right] & \text{if } \log\frac{L}{l} \leq p_1 \\[3em] p_0 + p_2\hat{y}(L)\exp\left[-\dfrac{\left(\log\frac{L}{l}-p_1\right)^2}{p_4}\right] & \text{if } \log\frac{L}{l} > p_1 \end{cases}$$

For hake juveniles and horse mackerel, we estimated $\hat{y}$ as the response prediction from a third degree polynomial beta regression model of the predator length, fitted on the observed consumption ratio y.

$\hat{y}(L) \equiv f(\hat{\eta})=1/[1 + \exp(-\hat{\eta})]$,

with

$\text{logit}(\mu) = \hat{\eta} \equiv \hat{\eta}(L,\hat{\beta})=\hat{\beta}_0 + L\hat{\beta}_1 + L^2\hat{\beta}_2+L^3\hat{\beta}_3$,
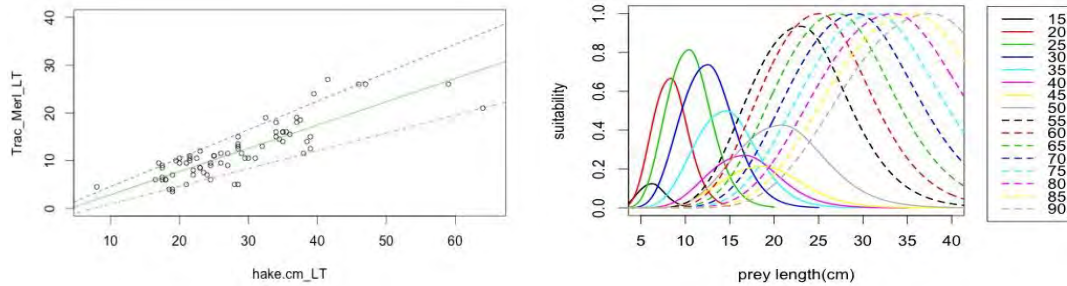
$y \sim BE(\mu, \phi)$.

Note that $\hat{y}$ only depend on L, as $\hat{\beta}$ is assumed to be known once the polynomial has been estimated out of GADGET.

Following the approach developed by Trenkel *et al.* (2004) we combined relationships between mean hake length and mean prey length (HOM, HKE, DPS) with quantile regression estimates (e.g Fig. 6.2.4 a) to shift the suitability function (Fig. 6.2.4 b) (by manipulating the p_1 parameter) until the predator size matched with the median prey size, as determined from observed data. The pread' of the suitability function was then manipulated (using the p_3 and p_4 parameters), until the 'tails' of the distribution coincided with the observed 10% and 90% quantiles observed in prey length – predator length relationships following the approach used by Trenkel *et al.* (2004).

The data used were prey length measures from hake stomach data collected in the study area in 2013-2014 (Fig. 6.2.4 a).

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Figure 6.2.4. a) Prey length – predator length relationship for hake (HKE) and horse mackerel (HOM) in the SoS area. b) Modified Andersen & Ursin (1977) suitability function, expressing the suitability of HKE and HOM in terms of length, for different HKE lengths.

The suitability function for consumption of DPS was still based on the proposed modified AU function, but due to few observed data, it was not based on quantile regression. Instead, we assumed that the DPS suitability for hake larger than 30 cm spans the range of the observed prey length (Fig. 6.2.5). Function is now created ad hoc to resemble the data variation:

$$\hat{y}(L) = \frac{1}{\left[1 + exp(-\sqrt{|L-30|})\right] * \left[1 + exp(-\sqrt{|L-35|}/1.5)\right]}$$
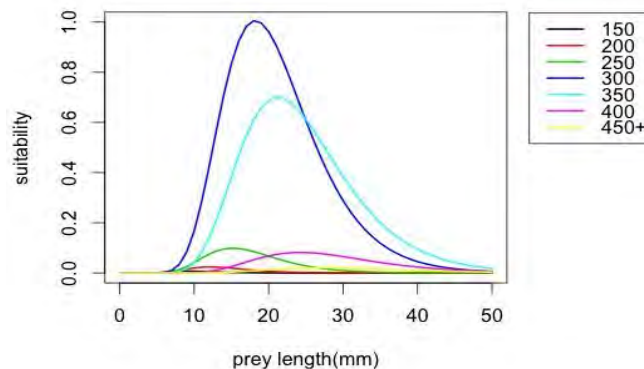
Figure 6.2.5. Modified Andersen & Ursin (1977) suitability function, expressing the suitability of DPS in terms of length, for different HKE lengths.

The datasets included in the GADGET SoS model and the relative contribution to the final total likelihood are showed in Table 6.2.1.

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Table 6.2.1. Likelihood components and their relative contribution to the final total likelihood (SSF:small-scale fishery)

Likelihood component	Description	Period	Relative weight
hake.aldist.commBMT	Hake age-length distributions from Italian trawlers	2005-2013	366.1
hake.aldist.commDP	Hake age-length distributions from Italian SSF	2005-2012	18.8
hake.ldist.commBMT	Hake length distributions from Italian trawlers	2005-2014	1388.2
hake.ldist.commDP	Hake length distributions from Italian SSF	2005-2014	16.4
hake.ldist.sur	Hake length distributions from Italian survey	2002-2014	452.6
hake.ldist.TUNcommBMT	Hake length distributions from Tunisian trawlers	2007-2014	501.7
hake.ldist.TUNcommDP	Hake length distributions from Tunisian SSF	2010-2014	13.2
pape.ldist.commBMT	Rose shrimp length distributions from Italian trawlers	2005-2014	31.6
pape.ldist.sur	Rose shrimp length distributions from Italian survey	2002-2014	34.8
pape.ldist.TUNcommBMT	Rose shrimp length distributions from Tunisian trawlers	2007-2014	44.4
trac.ldist.commBMT	Horse mackerel length distributions from Italian trawlers	2005-2014	56.9
trac.ldist.commDP	Horse mackerel length distributions from Italian SSF	2013	12.5
trac.ldist.sur	Horse mackerel length distributions from Italian survey	2002-2014	43.3
hake.sur.gp1	Hake abundance indices 0-20 cm from survey	2002-2014	23.8
hake.sur.gp2	Hake abundance indices 20-30 cm from survey	2002-2014	0.8
hake.sur.gp3	Hake abundance indices 30-40 cm from survey	2002-2014	0.5
hake.sur.gp4	Hake abundance indices >40 cm from survey	2002-2014	0.1
pape.sur.gp1	Rose shrimp abundance indices 0-10 mm from survey	2002-2014	2.9
pape.sur.gp2	Rose shrimp abundance indices 10-20 mm from survey	2002-2014	0.4
pape.sur.gp3	Rose shrimp abundance indices >20 mm from survey	2002-2014	0.4
trac.sur.gp1	Horse mackerel abundance indices 0-10 cm from survey	2002-2014	0.1
trac.sur.gp2	Horse mackerel abundance indices 10-20 cm from survey	2002-2014	0.8
trac.sur.gp3	Horse mackerel abundance indices >20 cm from survey	2002-2014	0.4
understocking	Understocking		1
bound	Penalty		0.5

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5.2.3. Model fitting (best model)

The implementation of the Gadget multi-species model has implied the parametrization of single species models which were linked by trophic interactions as well as interactions with fleets in a second modelling step. For each input data set a specific likelihood function was used to compare the model output to the data during the estimation. A likelihood score was calculated for each likelihood component and a weighted sum of all the likelihood scores was then used to calculate an overall likelihood score as described in Taylor et al. (2007). Model selection was based on the value that minimized the minus log-likelihood and visual inspection criteria based on the following components:

- i) observed and fitted length distributions in the fleets catch;
- ii) observed and fitted length distributions in the MEDITS survey;
- iii) observed and fitted MEDITS survey CPUE;
- iv) residuals.

A full description of the single species and multispecies Gadget models for the SoS area is provided in D 5.3.
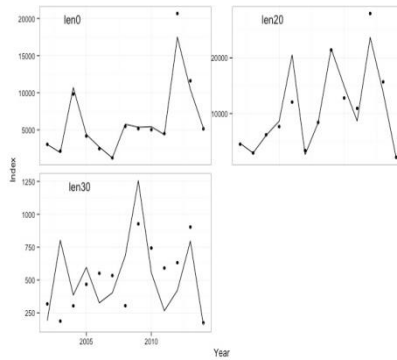
5.3. Results and discussion

Model fitting

In the next plots, the simulated values of both the single species and multispecies GADGET models are compared with the observed data for the period 2002-2014. Single species models appear able to produce reliable simulations of density indices (n km^{-2}) of the MEDITS bottow trawl survey (Fig. 6.3.1.). The only main inconsistency between observed and simulated data was for HKE > 40 cm TL.
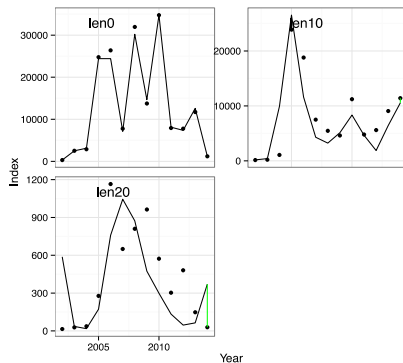
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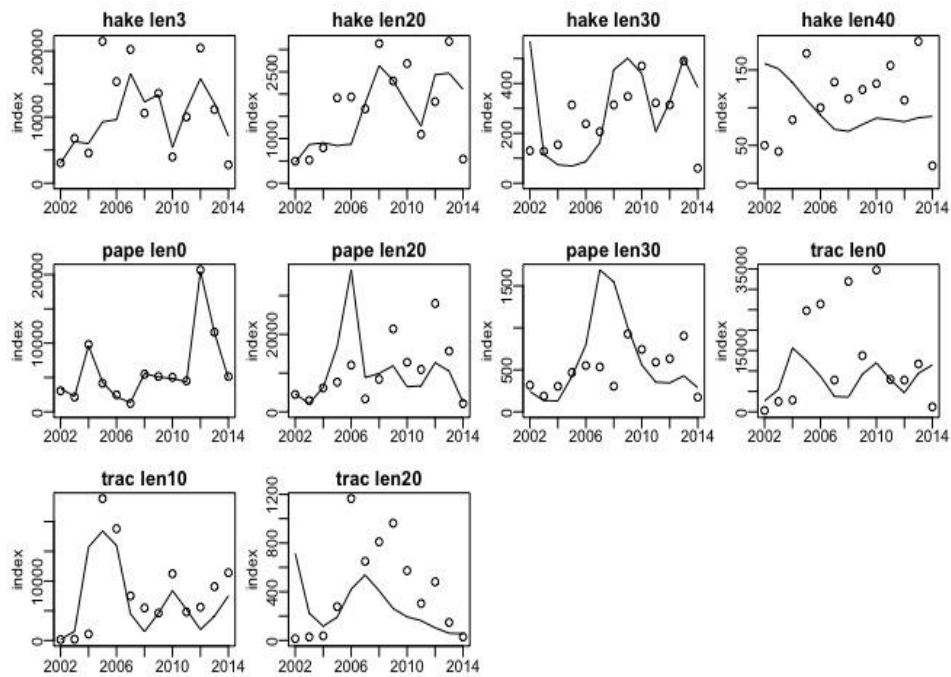
HKE
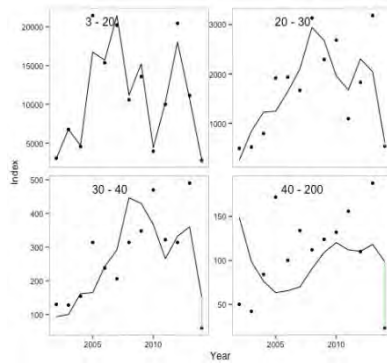
DPS

HOM

Figure 6.3.1. Single species GADGET model for hake (HKE), deep-water rose shrimp (DPS) and horse mackerel (HOM): observed (black dots) and estimated (solid line) density indices from the MEDITS trawl survey for different length groups. HKE: 3-20cm TL, 20-30 cm TL, 30-40 cm TL, >40 cm TL; DPS: 0-20 mm CL, 20-30 mm CL, >30 mm CL; HOM: 0-10 cm TL, 10-20 cm TL; > 20 cm TL

The multispecies Gadget is still able to produce reliable simulations of the MEDITS density indices although with less goodness of fit than single models ones (Fig. 6.3.2).

Figure 6.3.2. Multispecies GADGET model for hake (HKE), deep-water rose shrimp (DPS) and horse mackerel (HOM): observed (black dots) and estimated (solid line) abundance indices from the MEDITS trawl survey for different length groups (as in Fig. 6.3.1).
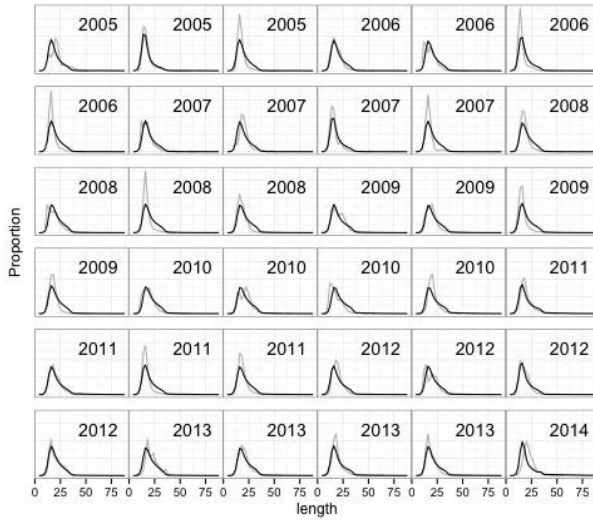
Single species Gadget models simulate consistently the quarterly length frequency distributions of the trawl catch of the Italian and Tunisian fleets of hake and rose shrimp, whereas it still does not simulate properly the horse mackerel catch structures (Fig. 6.3.3)

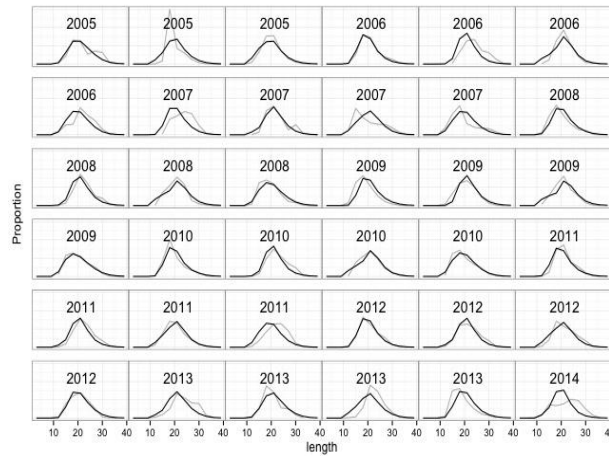## SINGLE SPECIES MODELS

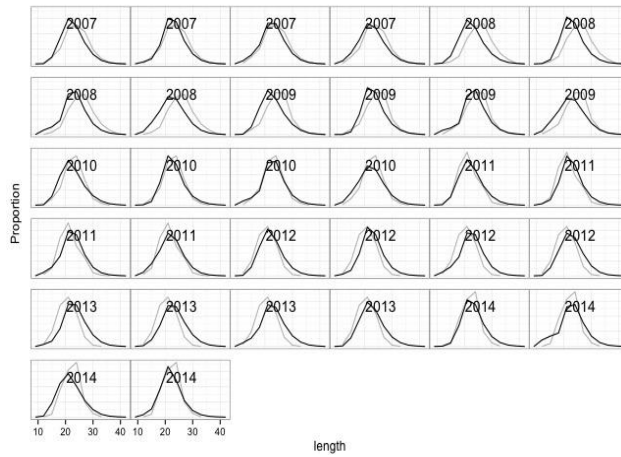### Italian trawlers (HKE)

Tunisian trawlers (HKE)

![MareFrame]

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Italian trawlers (DPS)

Tunisian trawlers (DPS)

![MareFrame logo]

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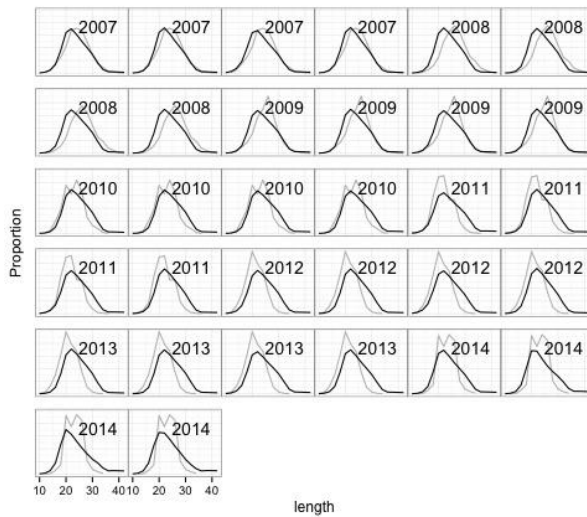
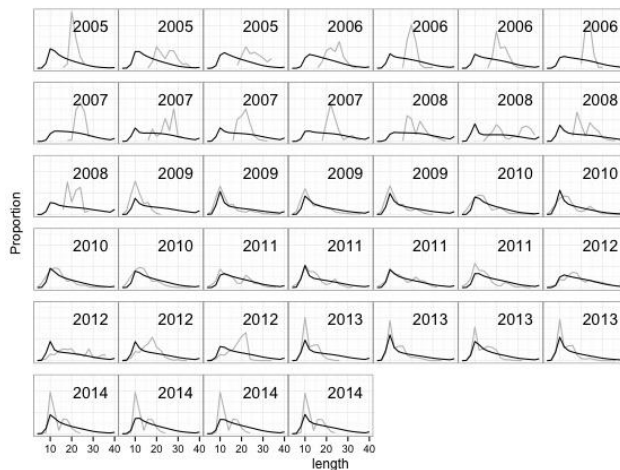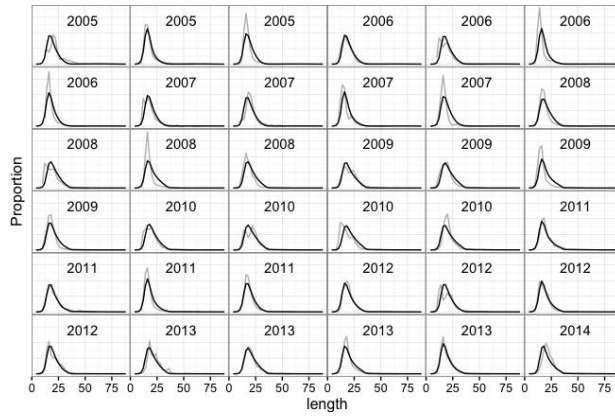**Italian trawlers (HOM)**

Tunisian trawlers (HOM)

Figure 6.3.3. Single species GADGET models for hake (HKE), deep-water rose shrimp (DPS) and horse mackerel (HOM): observed (grey lines) and simulated (black lines) length frequency distribution of the Italian and Tunisian trawlers.

In the multispecies configuration Gadget still produces reliable simulations of the catch structures of hake and deep-water rose shrimp landed by the Italian and Tunisian fleets (Fig. 6.3.4).
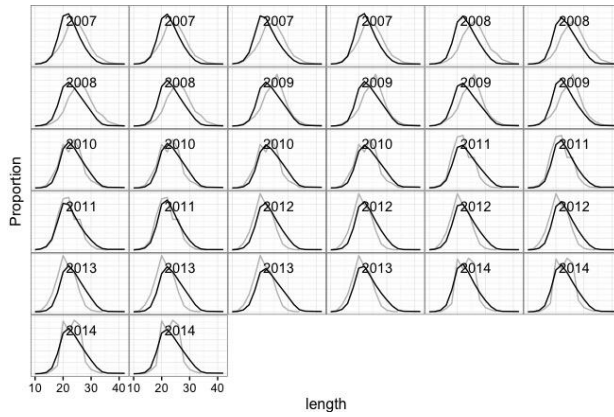
MULTISPECIES MODELS
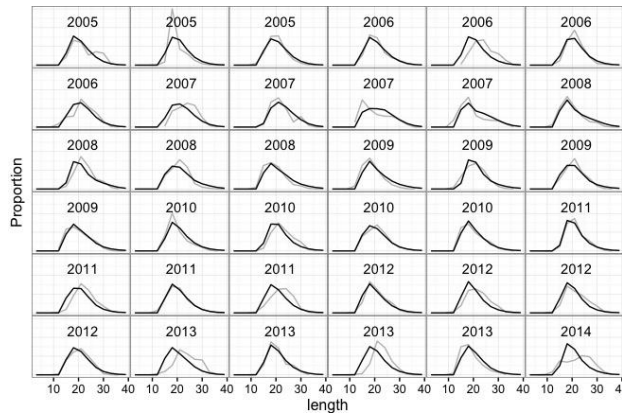
Italian trawlers (HKE)
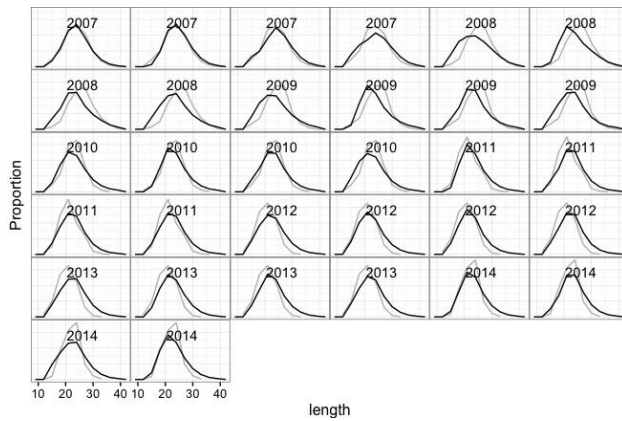

Tunisian trawlers (HKE)

Italian trawlers (DPS)

Tunisian trawlers (DPS)

Figure 6.3.4. Multispecies GADGET models for hake (HKE), deep-water rose shrimp (DPS) and horse mackerel (HOM): observed (grey lines) and simulated (black lines) length frequency distribution of the Italian and Tunisian trawlers.

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5.3.1. Indicators

GES indicators

The following GES indicators were calculated for hake and deep-water rose shrimp from the Gadget multispecies outputs (Fig. 6.3.5):

- spawning stock biomass (SSB), related to the reproductive capacity of the stock [MSFD: D3.2.1]
- fishing mortality (F), related to the level of pressure of fishing activity [MSFD: D3.1.1]

Gadget estimates of spawning stock biomass (SSB) and fishing mortality (F_{bar}) for HKE and DPS for the period 2002-2014 were compared with results of standard stock assessments using the Extended Survivor Analysis (XSA) for the period 2007-2014. Gadget produced SSB estimates for the two stocks that are comparable with XSA estimates. F_{bar} estimated by Gadget differs particularly at the beginning of the time series becoming more similar to the XSA in the last years (Fig. 6.3.5).

Hake
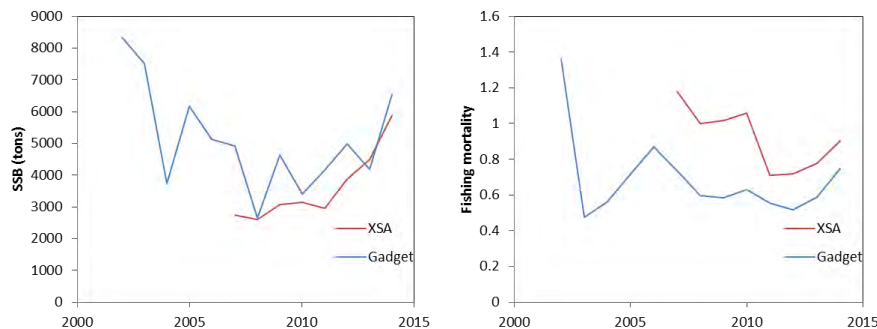
Deep-water rose shrimp
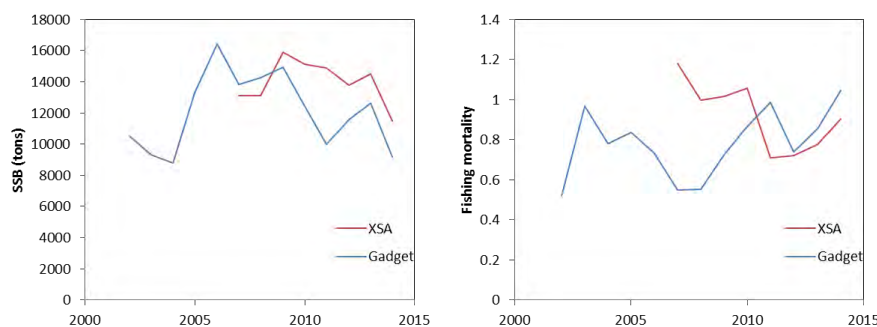
Figure 6.3.5. Multispecies Gadget model. Time series of model estimates of spawning stock biomass (SSB) and fishing mortality for hake (HKE, F_{1-6}) and deep-water rose shrimp (DPS, F_{1-3}) compared with standard assessment estimates from Extended Survivors Analysis (XSA, GFCM WGSAD, 2016).

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Socio-economic indicators

The fleet segments for which socio-economic indicators are calculated are the Italian trawlers 12-24 m LOA and and 24-40m LOA. The data used are official EU-DCR/DCF economic and transversal variables collected in GSA 16 (South of Sicily, Table 6.2.2).
The following indicators were adopted:
- Revenues (R)
- Total costs (C)
- Gross profits (P=R-C)
- Days-at-sea: assumed as an indicator correlated to the quality of life of fishers

Table 6.2.2 Economic and transversal variables of Italian bottom trawlers in GSA 16 obtained from the EU data collection.

	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
N. trawlers	507	514	518	467	478	455	465	430	392	382	388	391
GT TOT	35550	36468	38892	34801	35868	35524	36927	31400	30389	28905	28923	28756
KW TOT	122038	122768	129123	116853	122109	119113	122254	108072	104357	103163	104183	104611
Fishing days	81852.79	82556.93	89318.62	89163.66	78269.63	78585.65	78775.41	70539.09	63730.60	61156.36	54614.76	56166.53
Kw*days/1000000	9989.12	10135.33	11533.08	10419.00	9557.40	9360.54	9630.65	7623.29	6650.75	6309.07	5689.92	5875.63
Landings (t)	20800.2	21026.3	21227.6	20383.1	18134.5	18192.3	18914.3	17877.9	15286.3	13565.6	13225.3	14123.9
Oil consumption (l)	93227356	86645432	79207126	90822050	79369689	78953029	78435753	69475264	45571363	50493296	57201537	65248215
Oil costs (million €)	32.63	43.63	50.90	49.92	55.47	36.80	46.16	51.41	36.62	37.87	37.83	31.39
Labour costs (million €)	37.50	40.42	49.08	40.34	24.51	33.80	31.13	26.91	22.49	28.36	22.92	35.81
Other variable costs (million €)	8.50	10.84	13.10	13.33	11.29	11.40	11.57	10.45	7.08	6.83	3.79	4.22
Commercial costs (million €)	7.72	10.12	12.22	11.54	8.83	8.86	9.05	8.65	6.16	6.17	3.99	5.44
Maintainance costs (million €)	5.98	5.35	5.98	5.55	5.70	5.77	5.93	5.16	3.61	4.81	4.45	5.80
Fixed costs (million €)	6.31	6.85	7.56	6.86	7.05	6.97	7.15	6.18	4.96	4.70	3.79	3.95
Total costs (million €)	98.64	117.20	138.83	127.54	112.85	103.58	110.99	108.76	80.92	88.74	76.77	86.62
Revenues (million €)	136.15	159.25	189.31	170.20	134.49	134.46	137.83	132.88	110.19	102.88	93.84	117.69
Gross profits (million €)	37.51	42.05	50.48	42.66	21.64	30.88	26.85	24.12	29.27	14.14	17.06	31.07
Economic data for hake (HKE) and deep water rose shrimp (DPS)												
	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Land HKE (t) trawl	1949.2	1720.4	1597.7	1599.3	1367.6	1546.7	1519.3	1263.8	1393.2	1547.1	1385.8	1405.4
Land HKE (t) nets	61.2	69.6	28.6	119.0	27.5	34.7	18.4	20.7	31.5	4.3	81.8	205.7
Land DPS (t)	6665.0	8583.9	8441.1	5965.5	5941.0	7080.6	7699.9	7444.6	6081.9	5962.5	5310.4	6159.5
mean prize HKE (€/kg)	6.7	6.9	7.6	7.2	7.2	7.3	7.3	7.3	7.4	7.0	6.8	6.6
mean prize DPS (€/kg)	7.3	7.3	7.2	7.3	7.4	7.5	8.0	7.9	8.2	8.4	8.8	9.1
Gross revenues HKE (€/kg)	13.10	11.86	12.17	11.54	9.88	11.27	11.07	9.26	10.36	10.82	9.44	9.28
Gross revenues DPS (€/kg)	48.64	62.69	60.39	43.49	44.18	53.20	61.71	58.77	49.95	49.81	46.62	56.05

The proportion of costs related to the landings of DPS and HKE was calculated taking into account the relationships between total landings and the different costs associated. The revenues were obtained by multiplying the average selling price by the annual landings and, finally, gross profits were derived subtracting costs to the revenues.

The different costs associated to the fishing days at sea were obtained from linear regressions between these variables for the period 2004-2015. DPS mades up an average 40% of the total annual landing of the Italian trawl fleet in the period 2004-2015 whereas HKE was about 10% of

the total landing. DPS has therefore a key role for the economy of Italian trawlers whilst HKE can be considered as a commercial by catch of trawlers targeting DPS. Horse mackerel (HOM) is almost completely discarded at sea and its current commercial value is negligible. To simulate the effects of reducing fishing days on gross profits, the costs were considered as dependent by the number of days at sea only. In addition, the price of the two species were set as constant (i.e. geometric mean of the commercial price of 2013-2015) across the simulation period. The possible effects on profits related to change in the size composition of the landings (i.e. bigger shrimps have higher market price) as well as the price dynamics related to change in landings (i.e. the lower the landings the higher the commercial price) were not taken into account.

In the current configuration the multispecies Gadget model uses catch from the fleets exactly matching the landings data (i.e. the fleet declared as *total predator* by using total predator function). It was therefore not possible to compare Gadget simulations of fisheries catch and in turn economic indicators over the historical period covered by the model (2002-2016). The model was however used to forecast future catch and the associated costs, revenues and profits under different levels of catch reduction (i.e. business as usual and 20%, 50% reduction of the harvestable biomass) as showed in Fig. 6.3.6.
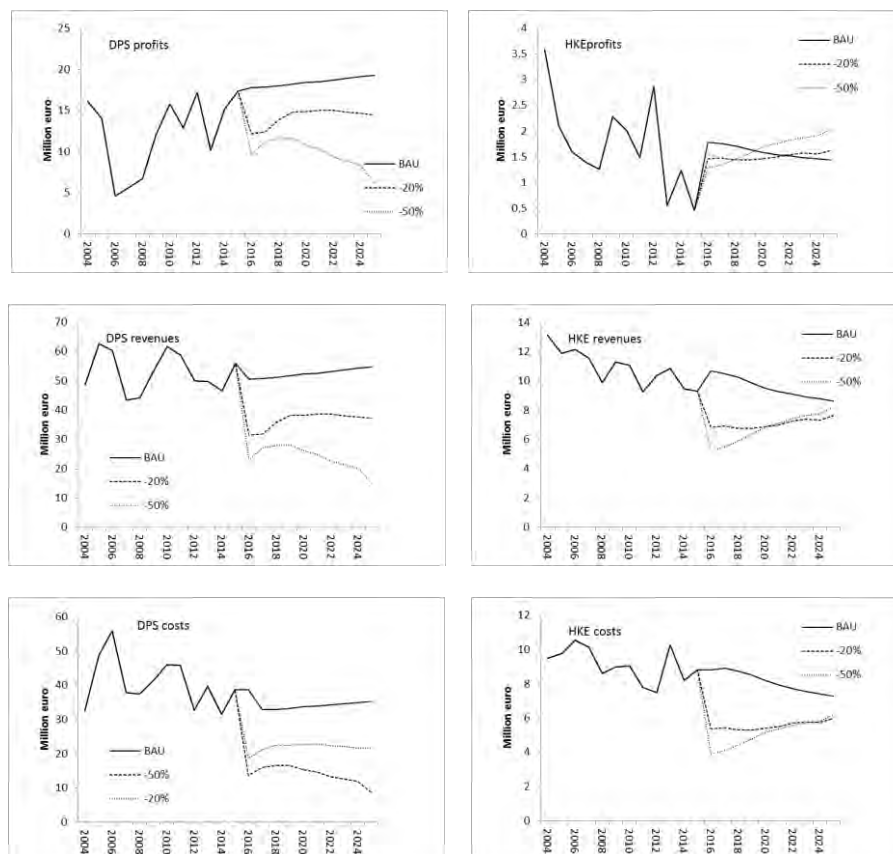
Figure 6.3.6. Multispecies Gadget model. Forecasted trend in gross profits, revenues and costs associated with the simulated catch of hake (HKE) and deep-water rose shrimp (DPS) under three different management scenarios: business as usual (BAU), -20% and -50% in the exploitation of the harvestable biomass.

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grant agreement no. 613571

5.4. Conclusion

A multispecies Gadget model has been developed and parameterized through the implementation of single species models for the main stocks targeted by off-shore trawl fisheries in the Strait of Sicily. It is the first multispecies Gadget model currently available for the Mediterranean and it will allow to explore the consequences of alternative management scenarios on the mixed fisheries targeting the deep-water rose shrimp (DPS) in the Strait of Sicily. This will be done taking into account the prey-predator relationships linking hake to DPS and horse mackerel. Results of different scenarios will be used to develop the prototype II of the Decision Support Tool.

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6. NORTH SEA CASE STUDY: ALTERNATIVES

6.1. Introduction

The North Sea Case study held its first Stakeholder Meeting on 14th May 2014. The main Stakeholder concerns were with

1. Need to achieve Fmsy
2. Landings Obligation
3. The Risks of Incompatible Regulations

They wanted a multispecies approach that would address these issues. This fitted well into the broader MAREFRAME aim of seeking to remove barriers that currently prevent a more widespread use of an ecosystem-based fisheries management.

For the North Sea the primary model is the GREEN MODEL (based upon Pope 1989 and Collie *et al* 2007) fitted to SMS results (Lewy and Vinther, 2004) and STECF EFFORT and ECONOMIC data sets(STECF,2016). The GREEN MODEL provides a front end model that the results of other models can be easily plugged into to provide social, economic and GES results. The secondary model developed for this deliverable is an EwE model (Mackinson & Daskalov, 2007) and recent model runs were kindly provided by the author (Mackinson pers comm). Like SMS this model has the virtue of having been extensively reviewed by the relevant ICES expert group over successive years (e.g. ICES, 2014). EwE has complementary strengths to SMS in that it is an end to end ecosystem model that considers a wider range of species and trophic levels than is possible to include in the core estimation process of SMS that is based upon a full assessment data set.

6.2. Methods

The Present North Sea Realisation of EwE considers 69 different species and functional groupings in its North Sea version. These are:

Baleen whales	Plaice
Toothed whales	Dab
Seals	Long-rough dab
Surface-feeding seabirds	Flounder
Juvenile sharks	Sole
Spurdog	Lemon sole
Large piscivorous sharks	Witch
Small sharks	Turbot
Juvenile rays	Megrim
Starry ray + others	Halibut
Thornback & Spotted ray	Dragonets
Skate + cuckoo ray	Catfish (Wolf-fish)
Cod (juvenile 0-2)	Large demersal fish

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Cod (adult)	Small demersal fish
Whiting (juvenile 0-1)	Miscellaneous filterfeeding pelagic fish
Whiting (adult)	Squid & cuttlefish
Haddock (juvenile 0-1)	Fish larvae
Haddock (adult)	Carnivorous zooplankton
	Herbivorous & Omnivorous zooplankton (copepods)
Saithe (juvenile 0-3)	
Saithe (adult)	Gelatinous zooplankton
Hake	Large crabs
Blue whiting	Nephrops
	Epifaunal macrobenthos (mobile grazers)
Norway pout	
Other gadoids (large)	Infaunal macrobenthos
Other gadoids (small)	Shrimp
	Small mobile epifauna (swarming crustaceans)
Monkfish	
Gurnards	Small infauna (polychaetes)
Herring (juvenile 0-1)	Mackerel
Herring (adult)	Horse mackerel
Sprat	Sandeels

The catch, biomass (SSB where differentiated in the the above list e.g. adult cod) and fishing mortality taken as the ratio of catch to biomass (where possible as adult catch to adult biomass) were input into the Green Model for the years 1991 to 2015. This provides complementary social, economic and GES results using these outputs of EWE. How the GREEN MODEL provides the social and economic results has been described in the North Sea section of D4.5. The various environmental GES measures are functions of the fishing intensity on appropriate species groups.

12 Core species are used in the Green Model. In the figures and at places in the text they are refered to by 3 digit codes as follows:-.

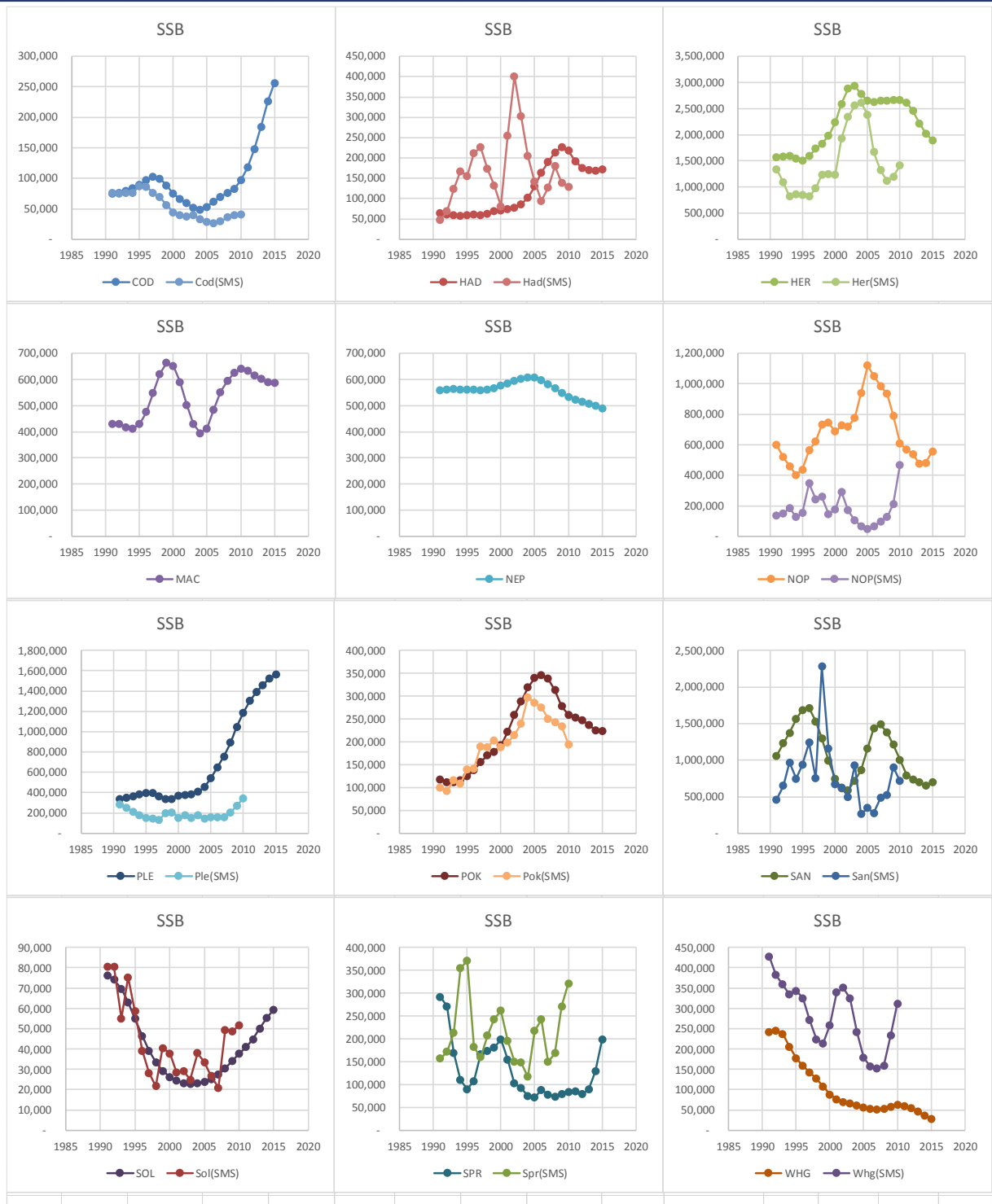| 3 digit code	Species English Name
Cod	Cod
Had	Haddock
Her	Herring
Mac	Mackerel
Nep	Nephrops
Nop	Norway pout
Ple	Plaice
Pok	Saithe
San	Sandeel
Sol	Sole
Spr	Sprat
Whg	Whiting

6.3. Results and Discussion

EwE provides results for biomass (in some cases Spawning Stock Biomass (SSB) in others total biomass) for the 12 core species that the Green Model (10 of which would in the primary model be SMS results) as shown in figure 7.1. species by species. These are compared to results from SMS for all but Mackerel and Nephrops. In general these results compare quite reasonably. There are however quite strong divergencies in the cases of haddock, plaice, Norway pout, sprat and whiting. In some cases this may be because EwE provides total biomass rather than SSB (e.g. plaice, Norway pout and sprat)).

Figure 7.2 shows the comparable catch figures from EwE. These are compared to catch results from the stock assessment graphs provided by the ICES website. All the ICES results are for landings when these are differentiated friom catch. Again these mostly match but there are discrepencies between results for haddock, plaice and whiting. In part these result from difference between landings and catch and some may result from different stock areas. This is because since some ICES stock data take in catch results from areas VIId and or VIa and/or IIIa as well as the North Sea *senso stricto* (areas IVa,b,c). Such differences may also be reflected in the discrepencies of biomass.

Figure 7.3 provides EwE estimates of fishing mortality rate. These are contrasted where available with SMS results. These compare quite reasonably for all stocks where the comparison is possible. All show a declining trend over at least the past 10 years reflecting the stronger management of the North Sea since about the year 2000.

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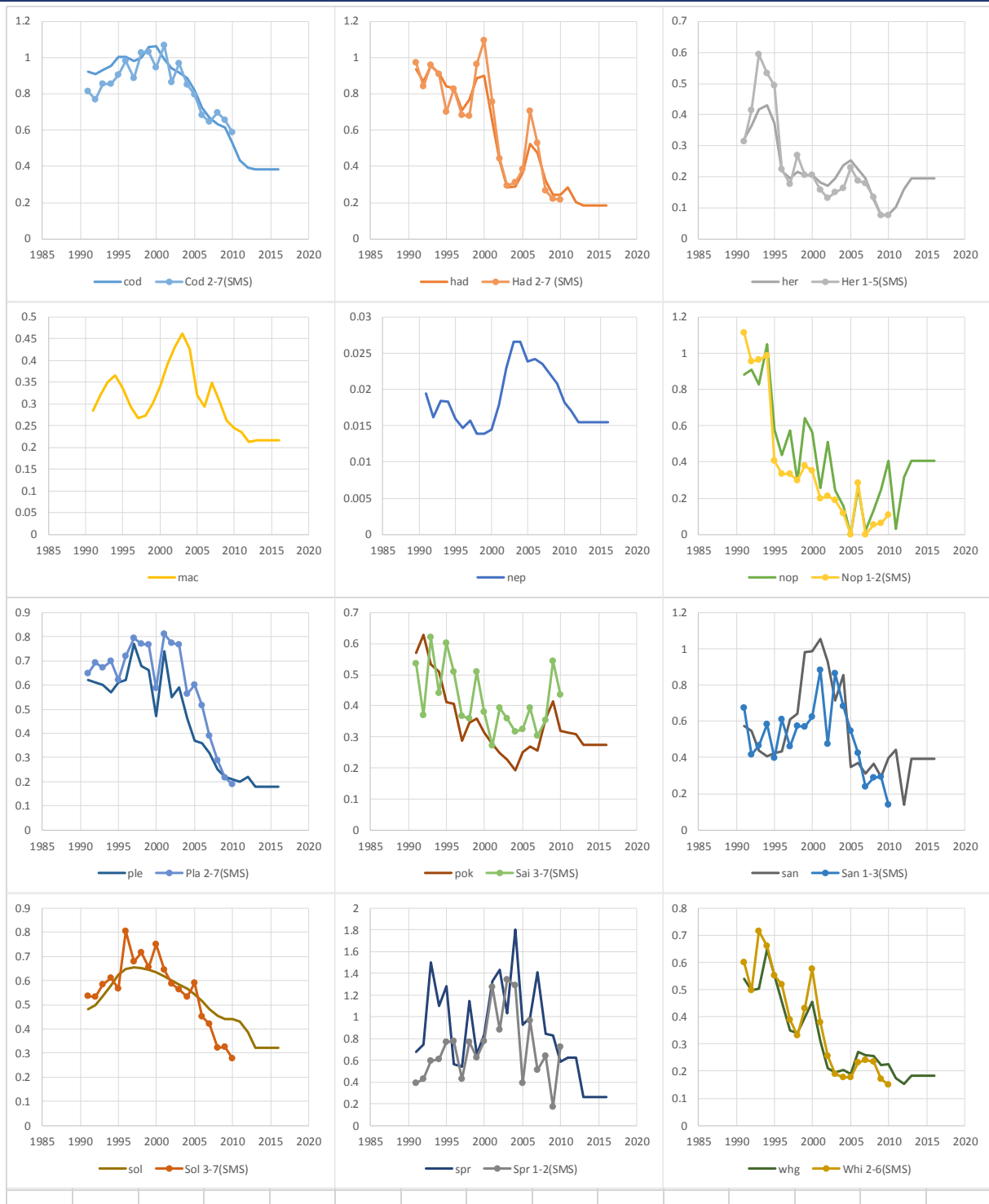
for the 12 core species

Figure 7.1. EwE provides results for biomass (in some cases Spawning Stock Biomass (SSB) in others total biomass) for the 12 core species that the Green Model (10 of which would in the primary model be SMS results) as shown in figure 7.1. species by species.
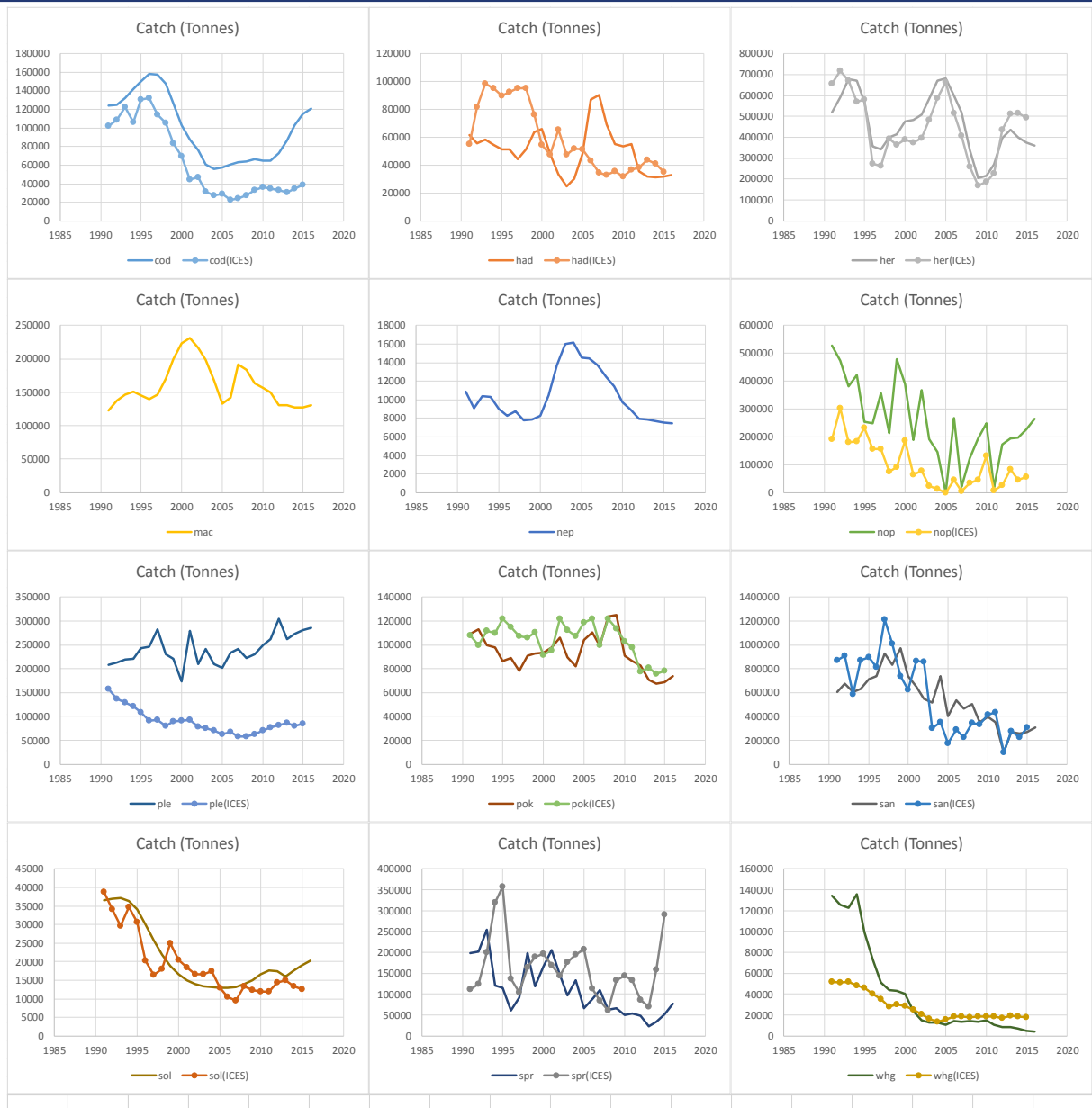
Figure 7.2 Comparison between catch for the 12 core species as estimated by EwE and results from the ICES website.

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Figure 7.3. Fishing Mortality Rate for the 12 core species from EwE and from SMS(where available).

In addition to the results for the core species, EwE provides biomass results for all ecosystem components. Figures 7.4, 7.5, 7.6, 7.7 and 7.8 respectively show the biomass results for Marine Mammals and sea birds, for elasmobranchs, for other roundfish, for other pelagic and flatfish and for other non core species groupings. It is a strength of EwE that these species are considered within the model rather than as in SMS (for those considered) as external species based upon survey abundance.
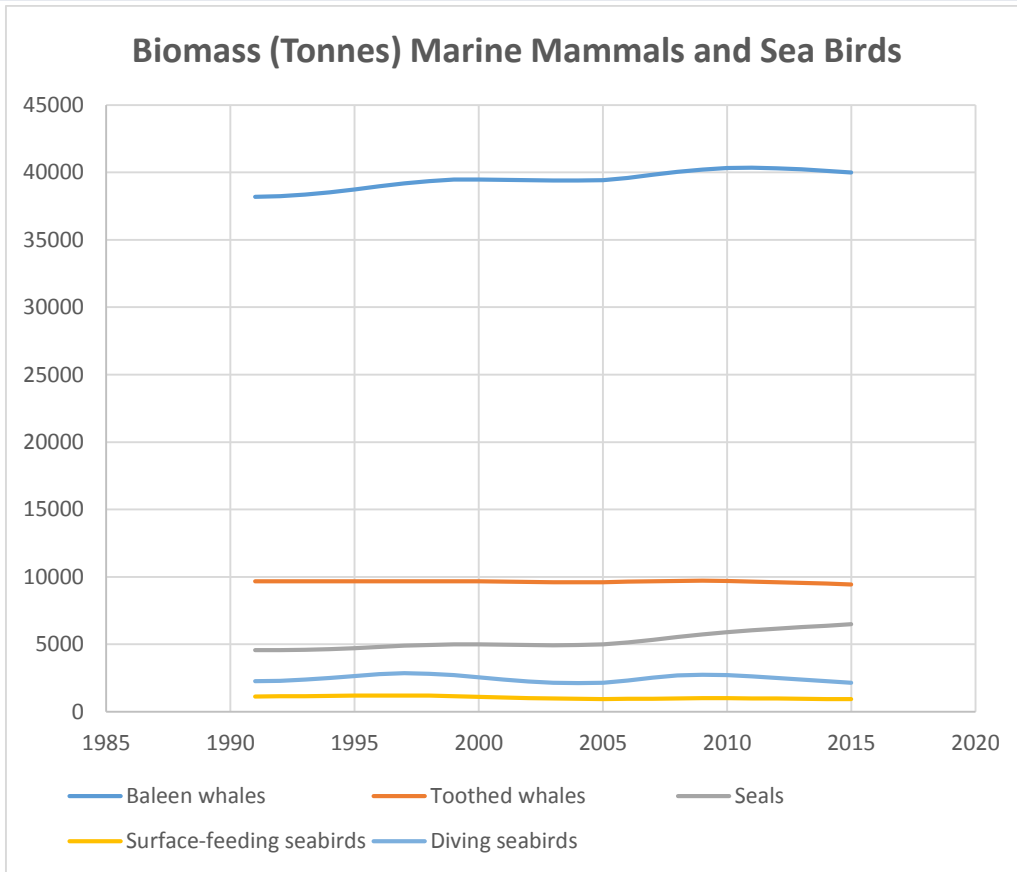
Figure 7.4. EwE biomass estimates for marine mammals and sea birds.
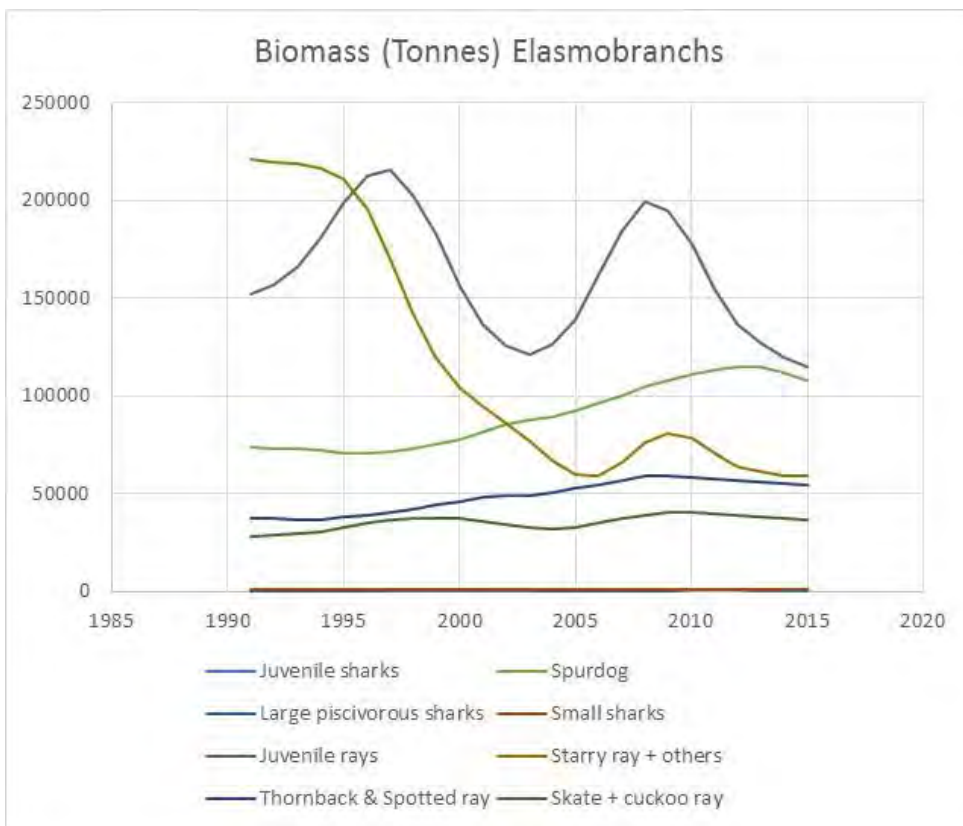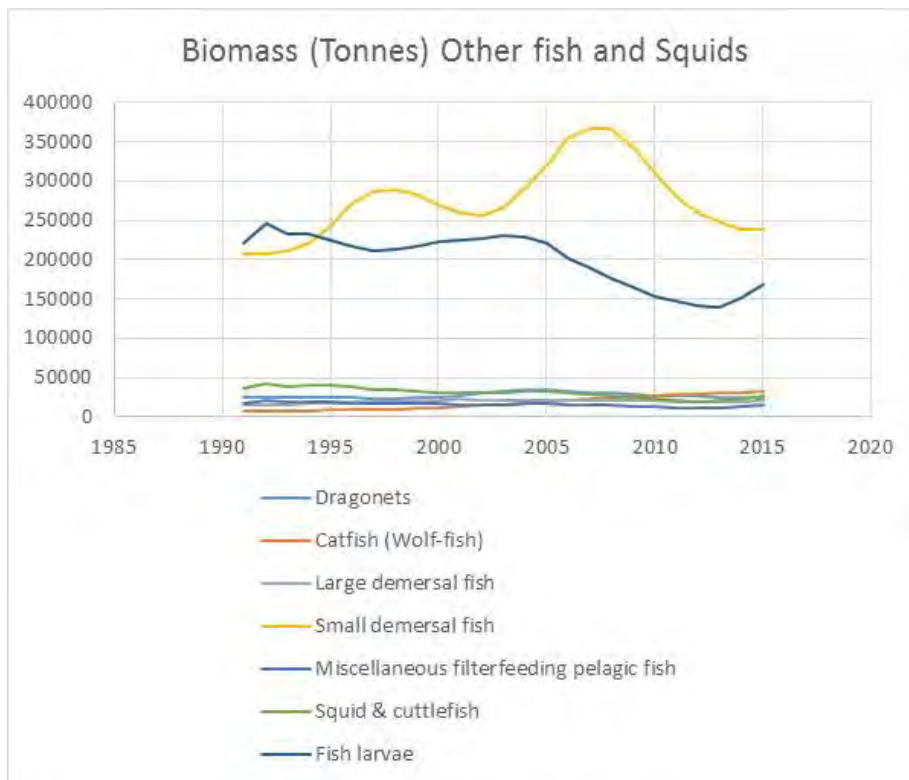

Figure 7.5. EwE biomasses for Elasmobranchs
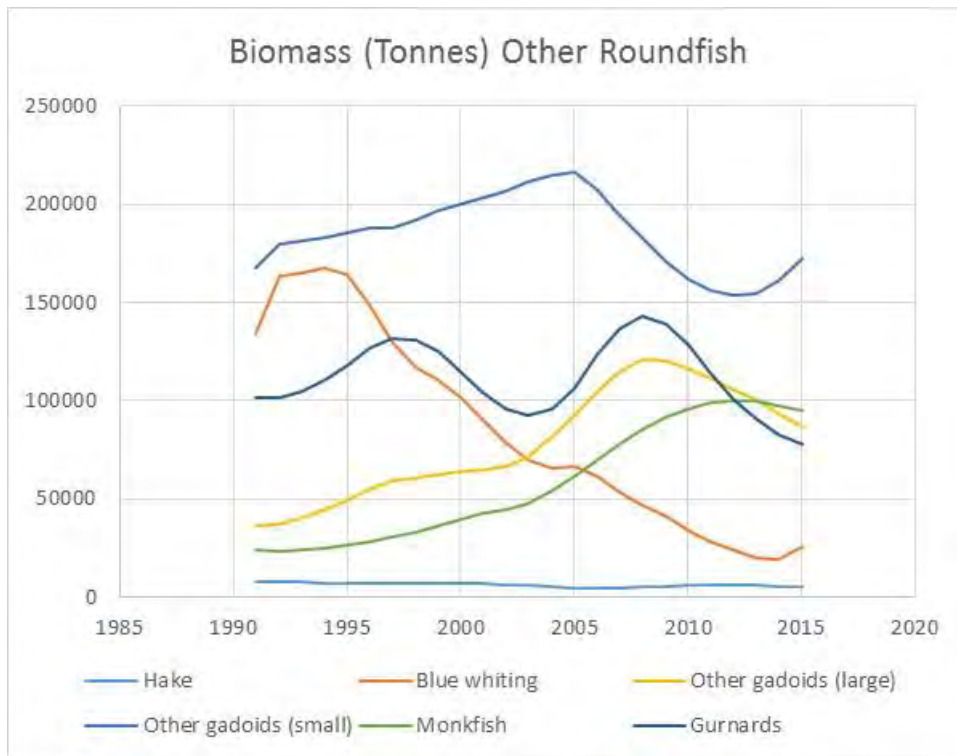
Figure 7.6. EwE Biomasses for other than core Roundfish species
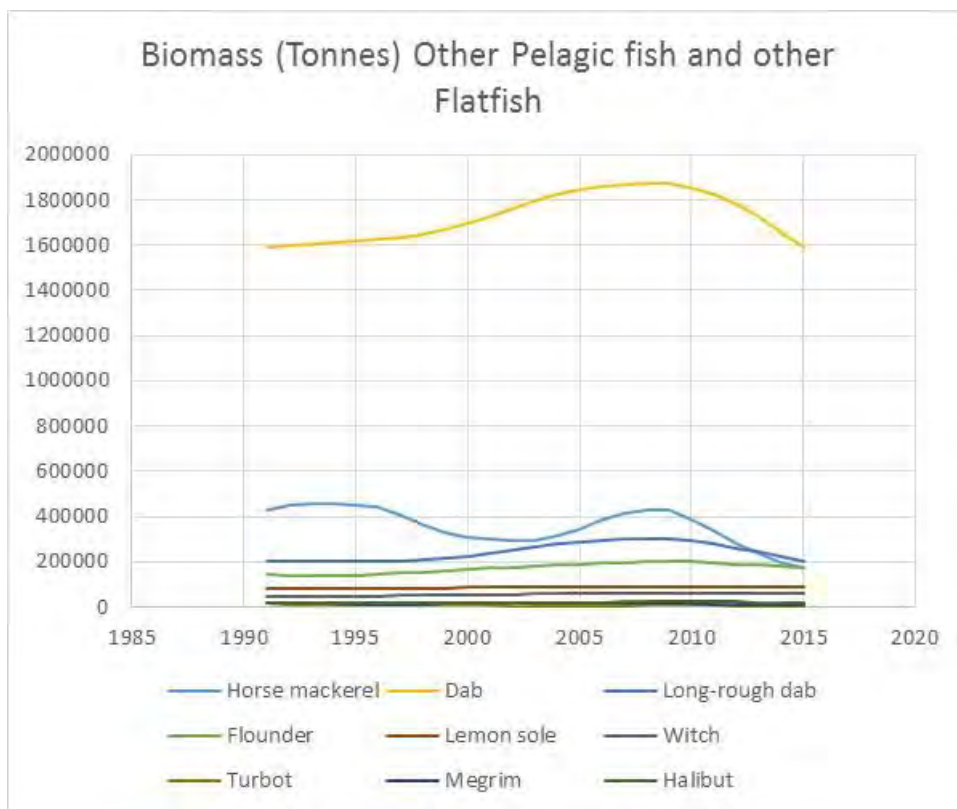
**Figure 7.7.** EwE Biomasses for other than core pelagic and flatfish species.

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Figure 7.8. EwE Biomasses for other species groupings.

Including the catch, biomass and fishing mortality results of EwE into the GREEN MODEL allows its standard social and economic outputs to be computed. Figure 7.9 shows the time series of the revenue, labour cost (the social measure), non labour costs, Gross Value added (GVA) a measure of the overall social benefit of the fishery, and gross profit obtained from the EwE inputs. These are compared in more recent years with results from the Green Models economic model calculated for D4.5. All results are in billion(10^9)€ at 2013 prices and costs and only represent EU fisheries (they exclude Norway).

In addition putting the EwE results into the GREEN MODEL provides the core fish stock GES measures. These may be seen in Figure 7.10 and relate to the number of the core stocks (excluding nephrops) that are above various key single species limits and target levels for SSB and fishing mortality rate. These all show an improving trend in at least the past 10 years. It should be noted that the fishing mortality target of Fmsy is probably neither realistic nor perhaps desirable in a multispecies fishery where predatory relationships exist.
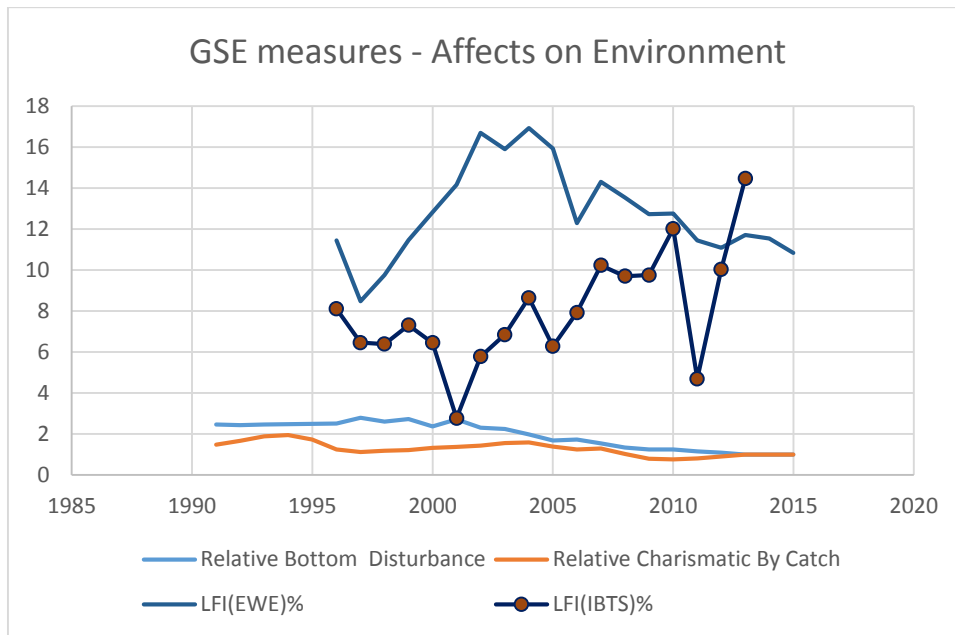
Figure 7.11shows GES measures, relative measures of bottom disturbance and charismatic by-catch potential relative to current levels of 1.0. These show a decline over the period studied that reflects the reductions in fishing mortality seen on all core North Sea species in recent years. It also shows measures of the Large Fish index (LFI) based upon an approximation that uses running averages of EwE estimates of fishing mortality rates for four L_∞ groupings of the core species. This is compared to the result obtained form all species from the International Bottom Trawl Survey of the North Sea. The general level of the LFI seems similar for these two series but the approximation does not follow the trend. This suggests that this approximation may need refinement.
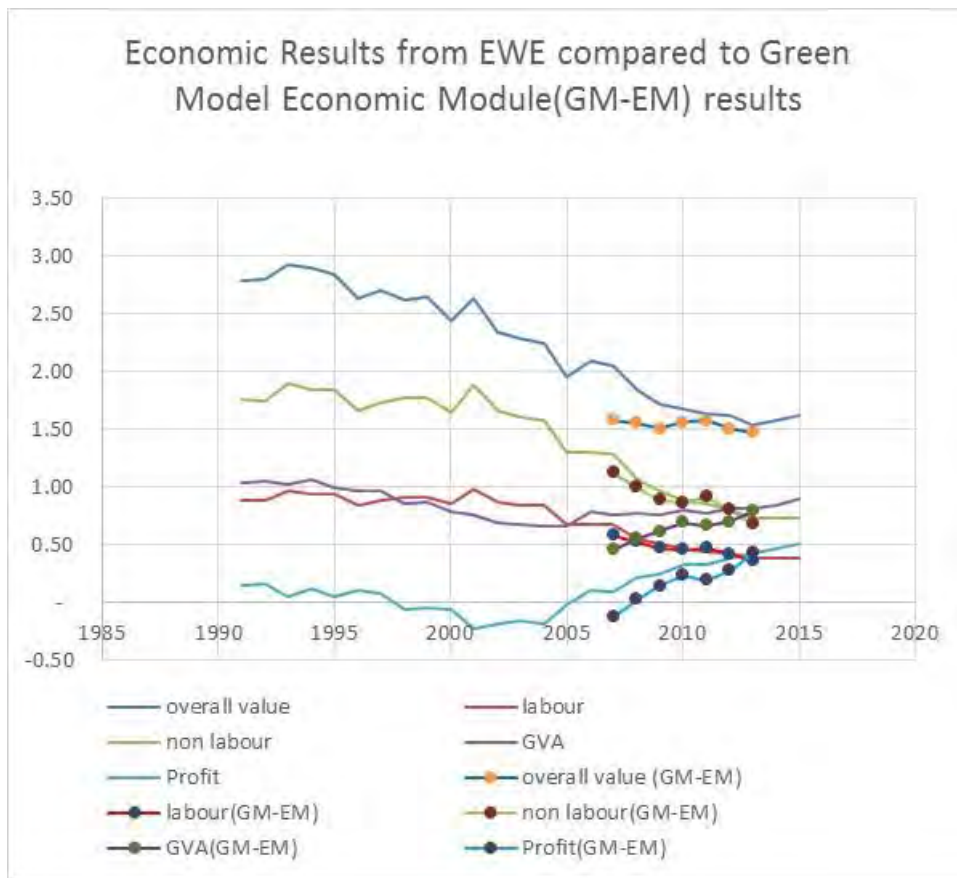
Figure 7.9 Social and Economic time series obtained by putting EwE results into the Green Model. These are contrasted with D4.5 results friom the Green Model. All values are in billion(109)€ (at 2013 prices and costs)and only refer to EU fisheries.
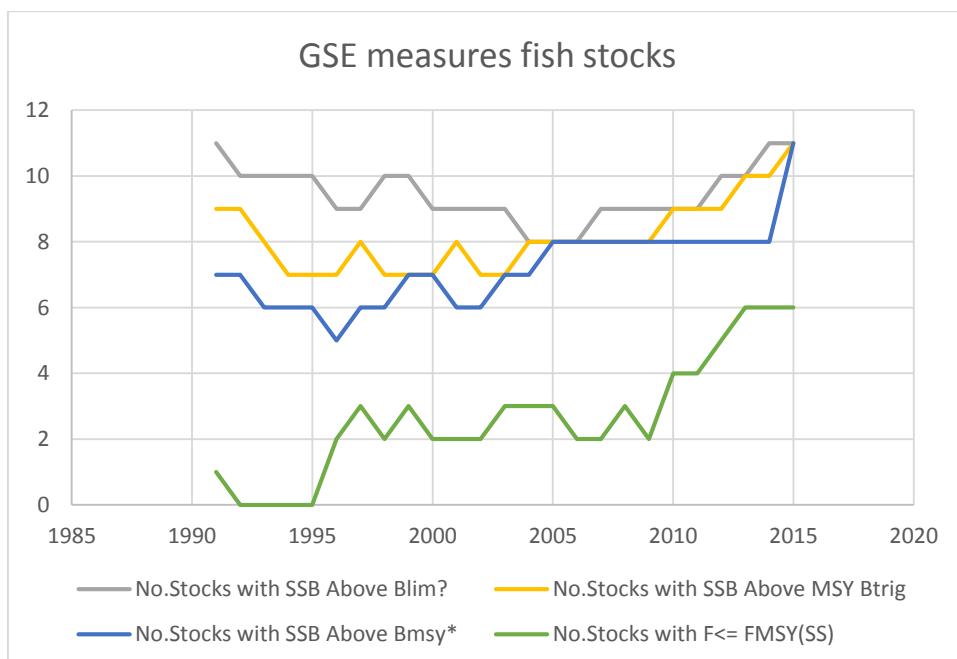
Figure 7.10 Core fish stock measures of GES obtained by adding EwE results to the Green Model.

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Figure 7.11 Environmental measures of GES obtained by adding EwE results to the Green Model.

6.4.   Conclusions

The EwE results provided from a recent run kindly made available by their author (Mackinson pers comm) compare well with the primary SMS model adopted for the North Sea. Bringing these results in to the Green Model additionally allows the social, economic and GES measures required by this deliverable to be calculated in a comparable way to results from SMS. This suggest that the strategy of providing a simple front end model such as the Green Model is a very useful way to compare and contrast the results of alternative models for the North Sea. Both SMS and EwE have the great virtue of having been carefully scruitinised by the relevant ICES Expert Group WGSAM (e.g. ICES 2014). This gives confidence as to their soundness and provides a test bed that other proposed North Sea multispecies models e.g the Thorpe Ensemble model (Thorpe *et al* 2015, 2016), the CSM or enhanced MS Schaefer models can be compared to in due course. Moreover the availablility of the green model will facilitate these comparisons.

6.5.   References

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7. SOUTH WESTERN WATERS CASE STUDY: GADGET

7.1. Introduction

This is the proposed second model for the SWW-IP Case of Study. It is a Ecopath with Ecosym (EwE) model for the Cantabrian Sea. Our first ecosystem model developed with GADGET considered the interaction between the fishery and cetaceans, with hake as a main commercial specie and pelagic species as other food This a MICES model (Models of Intermediate Complexity in Ecosystems.). The new model extends the ecosystem to 32 functional groups from lower (plankton) to upper trophic levels (cetaceans) which interact with the fishery. The EwE model will allow us to explore more complex trophic interactions in the ecosystem meanwhile the Gadget model will allow us to explore in more detail (size structure) the interactions among fishery, predators (cetaceans) and hake. Both models combined will provide a more complex view of how fishery and ecosystem interplays. As a first step towards a more complete analysis, the results of the EwE model in the Cantabric Sea are presented here.

The Cantabrian Sea (ICES area VIIIc) is located in a transition region between subtropical and boreal seas in the Eastern Atlantic, and fosters a complex ecosystem characterized by displaying higher biodiversity rates than adjacent areas (Olaso, 1990). The trophodynamic structure of the ecosystem were studied by Sanchez and Olaso (2004) using a mass- balanced Ecopath model based on the year 1994, finding out a complex food web architecture with well interconnected pelagic, demersal and benthic domains. Their model also pointed to the high impact of the fisheries in the ecosystem, which was placed among the most intense exploited temperate shelf ecosystems in the world. (Sanchez and Olaso, 2004).

We build on the aforementioned Ecopath model, splitting some of the invertebrate groups in mid- trophic levels and including seabirds and marine mammals as top predators, to obtain a more realistic representation of the coupling between the benthic and pelagic pathways. In addition, we also perform temporal simulations of the Ecopath model using Ecosim from the starting year 1994 to 2013. Along these 20 years the Cantabrian Sea Shelf Ecosystem has undergo important changes both in natural and anthropogenic pressures; the heavily exploited ecosystem from 1994 has experienced decreasing rates of fishing mortality which agree with the recorded reduction in the fleet sizes. On the other hand, the area is experiencing a progressive warming, which becomes more acute toward the inner Bay of Biscay and seems to be affecting the distribution of demersal and benthic species.

The Cantabrian Sea Shelf Ecosystem EwE model is structured on a empirically determined diet matrix, using 20 years of predator- prey data. These data were collected annually by stomach content sampling of the main fish predators at the IBTS Demersales surveys following a well established sampling methodology (see Velasco and Olaso, 1998, Preciado et al., 2005 or Lopez-Lopez et al., 2011 for a throughout description of sampling methodology). This time series of

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diet can be highly valuable for assessing the role of diet variability in ecosystem stability and resilience (as shown by Arroyo et al. under review).

In this report we propose the Ecopath and Ecosim model for the Cantabrian Sea Shelf Ecosystem as a tool for MAREFRAME. We describe the model and its parameterization, we discuss its main results and suggest an outline for future work.

7.2. Methods

Area of study

The study area, set in the Southern Bay of Biscay, encompasses the Cantabrian Sea and the northern part of the Galicia Shelf Sea, corresponding to the ICES division VIIIc. This area covers thus 21000 km2, including the continental shelf and upper slope (from 70 m depth up to 500 m depth approx.). This area has already been successfully modeled using Ecopath as a unit ecosystem (Sanchez and Olaso, 2004), notwithstanding the fact that several species have migratory habits and spend only part of their life in the study area.

The continental shelf on this area is characterized by its narrowness and, the patchy distribution of substrata and marked environmental gradients which result in a broad range of habitats (Serrano et al., 2006). The regional oceanography in the area varies seasonally, with the poleward current dominating during winter months and upwelling events taking over during the summer, mainly at the western edge of the study area (Gil, 2008; Alvarez et al., 2011). Spring and autumn are considered transitional seasons dominated by mesoscale oceanographic features (Gil, 2008). The small rivers mouthing in the area have limited influence in the regional oceanography (Gil, 2008).

Historically, local fisheries had been operating in the Cantabrian Sea, but the fleet industrialization in the last half of the 20thcentury yielded record catches and drove the ecosystem to a heavily exploited scenario. The trend seems to have reverted recently, as the fishing effort in the area in the last two decades has decreased (based on the estimates of fishing mortality) and the effects of this decreasing pressure have been proved at the ecosystem level using the large fish indicator (LFI, Modica et al., 2014) and ecosystem network properties such as connectivity and interaction strength (Arroyo et al., under review), all of them indicating ecosystem recovery and increased ecosystem stability.

Global warming is also affecting the Cantabrian Sea ecosystem, with increased temperature in surface waters (Tasker et al., 2008) and in intermediate waters toward the inner corner of the Bay of Biscay (Gonzalez- Pola et al., 2005). This warming is affecting the demersal assemblage, which is experiencing a meridionalization increasing most species abundances and frequencies of occurrence, meaning that their distribution seems to be expanding (Punzón et al., 2016). This effect is mainly acknowledged in species inhabiting the continental slope, whose distribution's centre of gravity has been getting deeper during the last decades (Punzón et al., 2016).
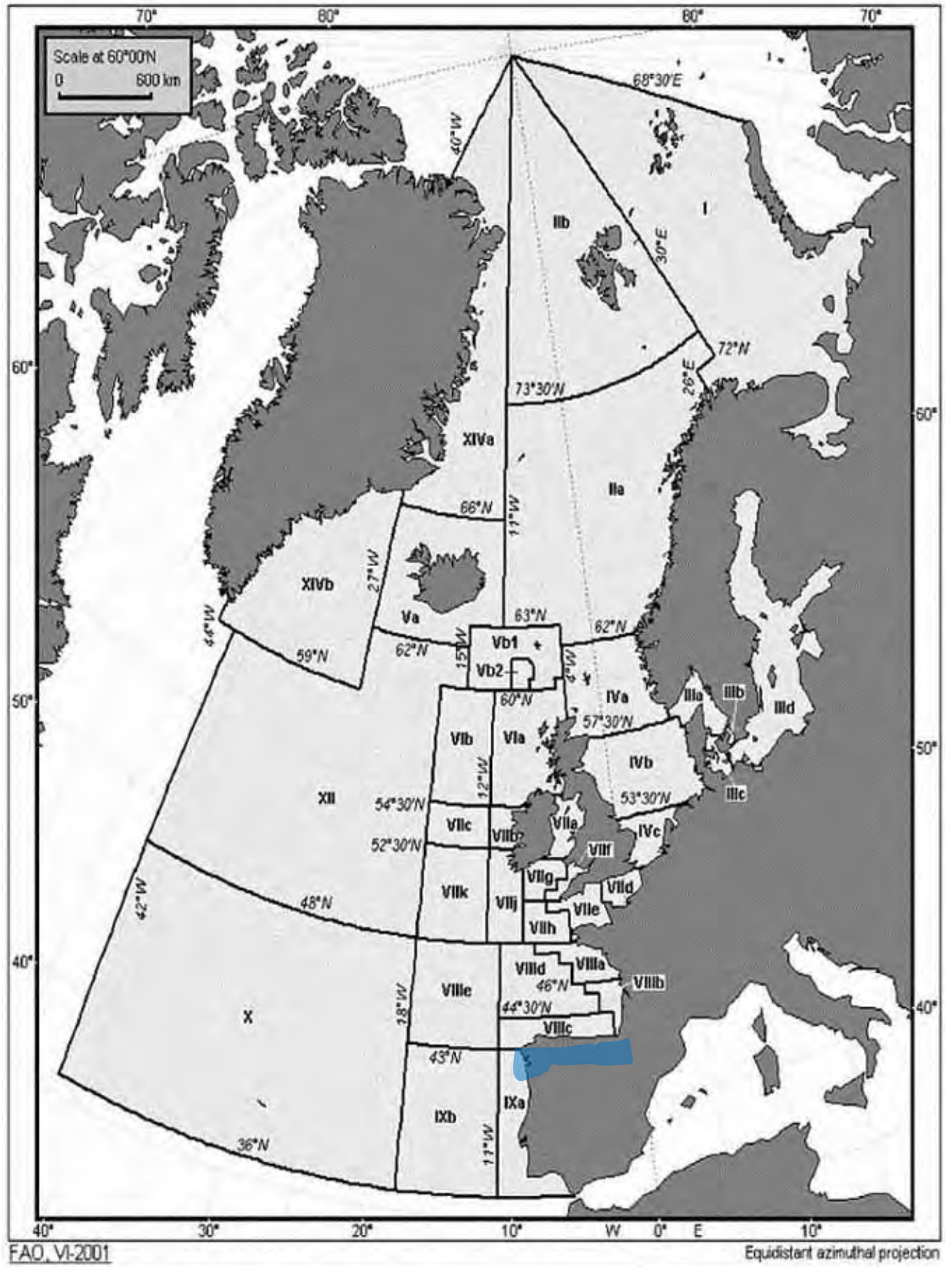
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Figure 8.1. Location of the modeled ecosystem: Cantabrian Sea (ICES area VIIIc).

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Ecopath: the mass-balanced model

Ecopath is based on two master equations, the first of which splits the production term for each functional group i into its components

$$P_i = Y_i + B_i \cdot M2_i + E_i + BA_i + M0_i$$

where, regarding group i, Pi is the total production rate, Yi is the total fisheries catch rate, M2i is the total predation rate, Bi the biomass, Ei the net migration rate (emigration- immigration), BAi the biomass accumulation rate and M0i is the so called "other mortality", which includes dying because of diseases, starvation, etc... or being consumed by predators not included in the model (M0i = Pi· (1- EEi,), where EEi is the ecotrophic efficiency of group i). The second master equation describes the energy balance within each functional group

$$Q_i = P_i + R_i + U_i$$

where, once again regarding group i, Qi stands for consumption, Ri for respiration and Ui for unassimilated food. Under these equations, the model is mass-balanced, and Ecopath uses a linear system of equations to estimate the missing parameters.

Our model is partly based on the pioneering work of Sanchez and Olaso (2004), as modifications in functional groups were needed to pursue the objectives of this work. In addition, after an intensive literature search most input data have been modified to meet the latest updates in the field (see Annex 8.1 and related literature). Nevertheless, the starting year for the model has been kept to 1994 due to the existence of a discard sampling program during this year, which covered the most important fleets in the area (Pérez et al., 1995). The study period covers 21 years, starting in 1994 and finishing in 2013, the last year for which data are readily available.

The model was constructed using the EwE (Ecosim & Ecopath) software version 6 (Christensen et al., 2008). We used the Ecopath routine to build the static model using year 1994 as the starting year of the series. To do so, we identified 34 functional groups representing a compromise between common trophic guilds, life history characteristics and the amount of available information. As a result we identified 16 fish groups, 6 groups of benthic invertebrates, 7 groups of pelagic invertebrates, one marine mammals' group, one seabirds' group, one primary producer and one detritus. One group (the fish predator Hake) was split into an adult and juvenile stanzas, due to its importance for the fisheries in the area and its relevance in the ecosystem as top predator (see Annex 8.1).

The fishery in the model includes the four fleets targeting the modeled species, i.e. demersal trawl, gillnet, longline and purse seine.

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Table 8.1. Ecopath model input and estimates after mass- balance (in italics)

Group name	TL	B	P/B	Q/B	EE	P/Q
Dolphins	4.90	0.033	0.072	23.760	0.138	0.003
Sea birds	3.82	0.006	4.600	70.000	0.156	0.066
Anglerfish	4.93	0.200	0.860	3.220	0.966	0.267
Large Dem Fish	4.59	0.100	0.960	4.140	0.667	0.232
Dogfish	4.32	0.434	0.460	1.980	0.415	0.232
Large Hake	4.78	0.524	0.880	3.680	0.824	0.239
Megrim	4.49	0.174	1.040	5.450	0.841	0.191
Benthic sharks	4.27	0.090	0.960	5.370	0.753	0.179
Rays	4.27	0.090	0.300	3.900	0.334	0.077
Squids	4.52	0.120	1.658	7.500	0.781	0.221
Benthic Cephalop	4.30	0.241	1.540	6.000	0.979	0.257
Small Hake	4.58	0.090	4.350	16.751	0.959	0.260
Other benthic fish	3.65	0.675	1.740	7.140	0.916	0.244
Mackerel	3.83	15.238	0.575	5.680	0.183	0.101
Blue Whiting	4.05	7.000	1.200	5.610	0.946	0.214
Small Dem Fish	3.87	0.690	1.640	5.000	0.941	0.328
Horse Mackerel	3.68	4.524	1.196	5.400	0.962	0.221
Anchovy	3.32	1.448	1.580	9.510	0.510	0.166
Sardine	2.96	4.152	1.416	7.650	0.608	0.185
Other planktophagous fish	3.51	4.500	1.450	8.400	0.930	0.173
Pelagic crab	2.92	0.175	3.011	12.000	0.945	0.251
Zooplankton feeding shrimps	3.50	1.350	3.210	9.220	0.970	0.348
Benthos-feeder decapods	3.40	0.960	2.062	14.650	0.973	0.141
Detritus-feeder decapods	2.81	0.970	3.154	17.570	0.977	0.180
Polychaetes	2.83	2.825	2.530	11.530	0.901	0.219
Other Invert	2.47	4.340	2.500	6.500	0.992	0.385
Migrating macrozooplankton	2.55	6.289	20.510	50.940	0.950	0.403
Gelatinous zooplankton	2.87	5.038	13.870	50.480	0.440	0.275
Detritivorous supreabenthos	2.00	3.446	15.620	50.000	0.950	0.312
Macrozooplankton	3.04	3.792	20.410	50.940	0.950	0.401
Mesozooplankton	2.03	16.010	25.000	90.400	0.998	0.277
Phytoplankton	1.00	11.300	262.800	0.000	0.475	
Discards	1.00	1.316			0.984	
Detritus	1.00	147.940			0.188	

The Ecopath model requires a large number of input data, i.e. at least four of the five following parameters for each functional group: the standing biomass of the group (B), its production (P/B; elaboration of new tissue), consumption (Q/B; intake of food), and ecotrophic efficiency (EE, the fraction of the group's biomass which is reutilized in the system). We provided B, P/B and Q/B for most functional groups (Table 8.1), letting the model estimate the ecotrophic efficiency, with exception of four groups for which biomass estimates were not available (i.e. Migrating macrozooplankton, Gelatinous zooplankton, Detritivorous suprabenthos and Macrozooplankton) whose EE was provided (Table 8.1) in order to estimate the group's biomass.

In addition to the former information, the model requires a diet matrix, which structures the ecosystem. Diet of most fish functional groups are empirically determined using the stomach content database at the IEO (Annex 8.1) and the input diet matrix for Ecopath was constructed

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using the year 1994 as a reference, whenever there was diet information available for this starting year. In functional groups comprised by several species the diet was weighted by the relative biomass of each species to conform the group's diet. When data for this initial year was not available or it was insufficient, we used the species mean diet along the study period. However, as the number of trophic links in which one functional group is involved remains invariable in the Ecosim simulations, we added some trophic interactions which were determined along the series although not specifically in the year 1994 (table S1). These interactions were only considered if the mean interaction strength along the series was > 0.01 % in terms of volume, and were included in the year 1994 with a minimum value of 0.01%. For several groups there was additional diet information, in addition to the IEO stomach content database. As an example, three sources of information were available for mackerel diet: information on its diet is available at the IEO database from stomach samplings in both bottom trawling and acoustic surveys and Bachiller (2012) determined mackerel diet in the eastern part of the Cantabrian Sea. When several sources of diet information were available we integrated them in the diet matrix. All sources used to construct the diet matrix can be consulted in Annex 8.1. The diet matrix also considers the relative time that a group is feeding outside the study area, incorporating the corresponding percentage of the diet as import. Species migrations in the study area, i.e. the annual migrations of mackerel, blue whiting and sardine, were accounted for in the diet matrix, assuming a fraction of its diet corresponds to the time the species was not present in the study area and thus is considered as imported.

Fishery landings were provided by the Information and Sampling Network of the Spanish Institute of Oceanography in the ICES VIII area, while fishery discards were estimated based on the discard sampling program carried out during 1994 (Pérez et al., 1995).

Model balancing was carried out manually following a top-down strategy, as data from higher trophic levels was generally more reliable (higher pedigree) and thus it needed less intervention. Inconsistent values were slightly modified following the criteria given by Christensen et al. (2008) to meet sensible data in the output parameters.

Biomass estimates directly obtained from assessment surveys are commonly too low to mass-balanced ecosystem models, thus a catchability correction factor was used to adjust the biomass estimates and meet the model mass requirements. It is acknowledge that some faunal groups are underestimated in trawl surveys (e.g. Sanchez and Olaso, 2004; Tsagarakis et al., 2010; Torres et al., 2013) depending on a series life history attributes such as species distribution, vertical migrations and body size among others. Biomasses were thus the firsts parameters modified during model balancing and P/B values were modified accordingly. Lastly, minor adjustments in the diet matrix were needed to achieve the final mass-balanced Ecopath model.

The resulting input data agreed with the prebalancing criteria by Link (2010), PREBAL, indicating the model follows some basic principles of ecosystem ecology, i.e. the resulting biomass data spanned 5 orders of magnitude from the least to the most abundant functional group, Biomass (B) declined by 5-10% on a log scale along the groups ranked by trophic level and Production/Biomass (P/B) and Consumption/Biomass (Q/B) also showed similar declines (Fig S1)
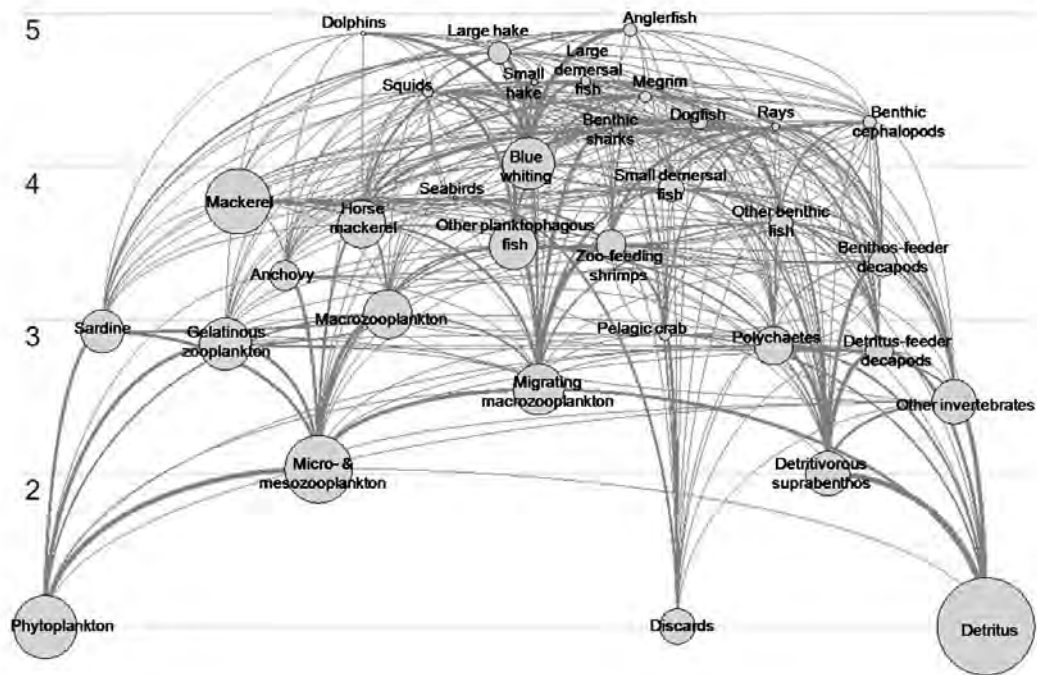
Figure 8.2. Flow diagram defining the structure of the Cantabrian Sea Shelf Ecopath model. The scale on the left corresponds to trophic level and circles are scaled to the group's biomass. Generally, a gradient from pelagic to benthic main pathway can be indentified reading the figure from left to right.

Ecosim: the temporal- dynamic model

Ecosim uses the system of equations on which Ecopath is structured to estimate biomass fluxes based on a system of differential equations as follows:

$$\frac{dB_i}{dt} = g_i \cdot \sum_j Q_{ij} - \sum_j Q_{ij} + I_i - (M0_i + F_i + E_i) \cdot B_i$$

where, regarding the group i, dBi/dt is the biomass growth rate during the time interval dt, Ii is the immigration rate, M0i is the natural mortality, Fi the fishing mortality, Ei the emigration rate and Qij is the consumption rate by each of its predators j. These consumption rates are calculated in Ecosim after the "foraging arena" theory, assuming that the prey biomass (Bi) is divided into a vulnerable and an invulnerable fraction, to account for behavioural strategies of the prey to balance predation risk with foraging (Ahrens et al., 2012). The vulnerable fraction is calculated by attributing a vulnerability (vij) to each predator-prey interaction, which biases the relation towards a bottom-up control (vij ≈ 1) or top-down control (vij ≥ 100). The default value in Ecosim describes an intermediate effect (vij=2).

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The model was parameterised for the years 1994- 2013, aiming at reliably reproducing the historical data. For this time period, biomass estimates and landings of several functional groups were available and were used to valid the generated outcome.

During the fitting procedure several alternative models were evaluated and compared, which essentially differed in the combinations of drivers considered. This fitting method has been successfully applied several EwE models (Mackinson et al, 2009; Tomczak et al., 2012; Alenxander et al., 2014). In the Cantabrian Sea we considered three main drivers of the ecosystem: trophic effects, primary production and fishing effort. The two firsts are determined using the "fit to timeseries" routine in Ecosim, which, in the case of trophic effects adjust the most sensitive vulnerabilities in order to reduce the sum of squares (SS) of the model and when calculating the primary production anomaly identifies a the pattern of a forcing function along the time series which applies to the primary production input of Ecopath to reduce the model SS. Relative fishing effort, on the other hand, must be introduced as a forcing time series for each fleet, with the Ecopath landings and discards (for the initial year) corresponding to the unit fisheries effort.

The fisheries effort series were provided by the Information and Sampling Network of the Spanish Institute of Oceanography in the ICES VIII area, using fishing days as the common currency. Particularly, the effort data of the trawling fleet for this area covers our study period and no further modification was needed. However, effort data for the other fleets covered a shorter period and we had to use back calculate the first years of the series based on the available information, as EwE would not allow a forcing time series with missing data (Christensen et al., 2008). We used the fleet size as provided by the Ministry of Agriculture, Food and Environment in the List of Registered Vessels as a proxy for estimating the annual fishing activity. In this census, the vessels are categorized by their fishing gear, with exception of one category of vessels small scale fisheries "artes menores" which is allowed to shift gears as it pleases. The latter is thus not included in the fleet size, and thus numbers might be slightly underestimated, particularly for the gillnet and longline fisheries.

The candidate models tested covered all possible combinations of drivers as follows:

1. Baseline model: Projection of the Ecopath model along the modeled time period with no additional driver.
2. Trophic effects only: Optimal 5 vulnerabilities were identified using the "fit to time series" routine and added to the baseline model as the only driver.
3. Trophic effects only: Optimal 20 vulnerabilities were identified using the "fit to time series" routine and added to the baseline model as the only driver.
4. Fishing effects only: The fishing effort series were added as the only driver to the baseline model.
5. Primary production only: The primary production anomaly was identified using the "fit to time series" routine and applied to the baseline model as the only driver.
6. Trophic effects+ primary production: The optimal vulnerabilities and the primary production anomaly were identified using the "fit to time series" routine and applied to the model.

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7. Trophic effects + fishing effort: The optimal vulnerabilities were identified using the "fit to time series" routine and applied to the model together with the fishing effort time series.

8. Primary production + fishing effort: The primary production anomaly was identified using the "fit to time series" routine and applied to the model together with the fishing effort time series.

9. Trophic effects + primary production + fishing effort: The optimal vulnerabilities and the primary production anomaly were identified using the "fit to time series" routine and applied to the model together with the fishing effort.

The model 's goodness of fit (SS) were compared and the Akaike Information Criterion was used to select the best model (Mackinson et al., 2009). AIC provides a mean for model selection as it measures the relative quality of a set of models based on the goodness of fit but penalizing the models as the number of parameters increase.

The model output was used to calculate several ecosystem trend indicators based of the functional groups' trophic levels.

7.3. Results

The balanced model (Figure 8.2) represents a well developed and round ecosystem (relative high omnivory indices), mainly driven by primary production which mobilizes approximately 3/4 of the basal biomass, despite the large biomass of detritus available. The ecosystem gets structured in 5 trophic levels, with dolphis, large hake and anglerfish as top predators (Figure 8.2). The biomass of discards is almost fully consumed by several functional groups (Ecotrophy Efficiency ≈ 1, see Table 8.1).

The best model was that including both fishing effort and trophic effects as driver in Ecosim (Table 8.2, model candidate 8). The inclusion of a forcing function to simulate the temporal trends in primary production did not improve the model performance under any of the configurations. Indeed, we found a lack of correlation between the primary production anomaly and the time series, which explains why the inclusion of this forcing function did not improve the EwE model.

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Table 8.2. Fitting results of the 9 candidate model configurations tested with Ecosim, with v indicating the number of vulnerabilities adjusted and sp the number of splines allowed in the primary production forcing function. N is the number of observations, MinSS is the minimum sum of squares, K is the number of parameters, AIC is the Akaike Information Criterion and AICc includes a correction for the AIC to account for sample size and Improved fit (%) is the percentage of improvement of each candidate model compared with the baseline model (model 1).

Candidate models	N	MinSS	K	AIC	AICc	Improved fit (%)
1. Baseline	405	118.34	0	-216.4	-216.4	
2. Baseline + Trophic effects (5v)	405	117.04	5	-208.3	-208.2	-3.77
3. Baseline + Trophic effects (20v)	405	113.5	20	-183.7	-181.8	-16.00
4. Baseline + PP anomaly (3sp)	405	116.4	3	-213.3	-213.3	-1.44
5. Baseline + Fishing forcing	485	111.9	0	-308.9	-308.9	42.75
6. Baseline + PP anomaly + Trophic effects (5v + 3sp)	405	115,38	8	-204.9	-204.6	-5.47
7. Baseline + Fishing Forcing + PP (5 sp)	485	110.41	5	-301.7	-301.6	39.39
8. Baseline + Fishing forcing + Trophic effects (5v)	485	106.05	5	-310.2	-310.1	43.31
9. Baseline + Fishing forcing + PP + Trophic effects (6v +5sp)	485	108.1	11	-294.2	-293.7	35.73

The vulnerabilities were mainly kept at the default value since increasing the number of adjusted vulnerabilities did not increase the model's AIC. After a steep improvement in the model with the adjustment of the most sensitive predator- prey vulnerabilities(see table 8.3 for vulnerabilities adjusted in the final model), including additional vulnerabilities tended to decrease the AICc improvement due to the penalization of including more parameters (for an example, compare models 2 and 3 in table 8.2).

The selected model reproduced credibly the time series variations of biomass in groups with high trophic level (TL), such as anglerfish, megrim, benthic sharks (figure 8.3). The model had, however, limited ability to reproduce the variability of mid-TLs, possibly because these correspond mostly to short-lived species highly dependent on environmental conditions. As primary production anomaly was not included in the model, the final EwE model lacks variability at the lower trophic levels. In the particular case of anchovy, mackerel and blue whiting, which are seasonal migratory species into the Cantabrian Sea, interannual differences in the strength of the migration could be driving the high interannual variability in observations, which the model fails to reproduce.

Table 8.3. *Summary* table with the five adjusted vulnerabilities (shaded cells) in the final model (model *configuration* 8).

Prey \ predator	Anglerfish	Benthic sharks	Blue Whiting	Horse Mackerel
Blue Whiting	1.00	2.00	2.00	
Small Dem Fish	2.00	2.00	2.00	
Horse Mackerel	1.01	35.59	1.24	
Anchovy		2.00	2.00	100.00

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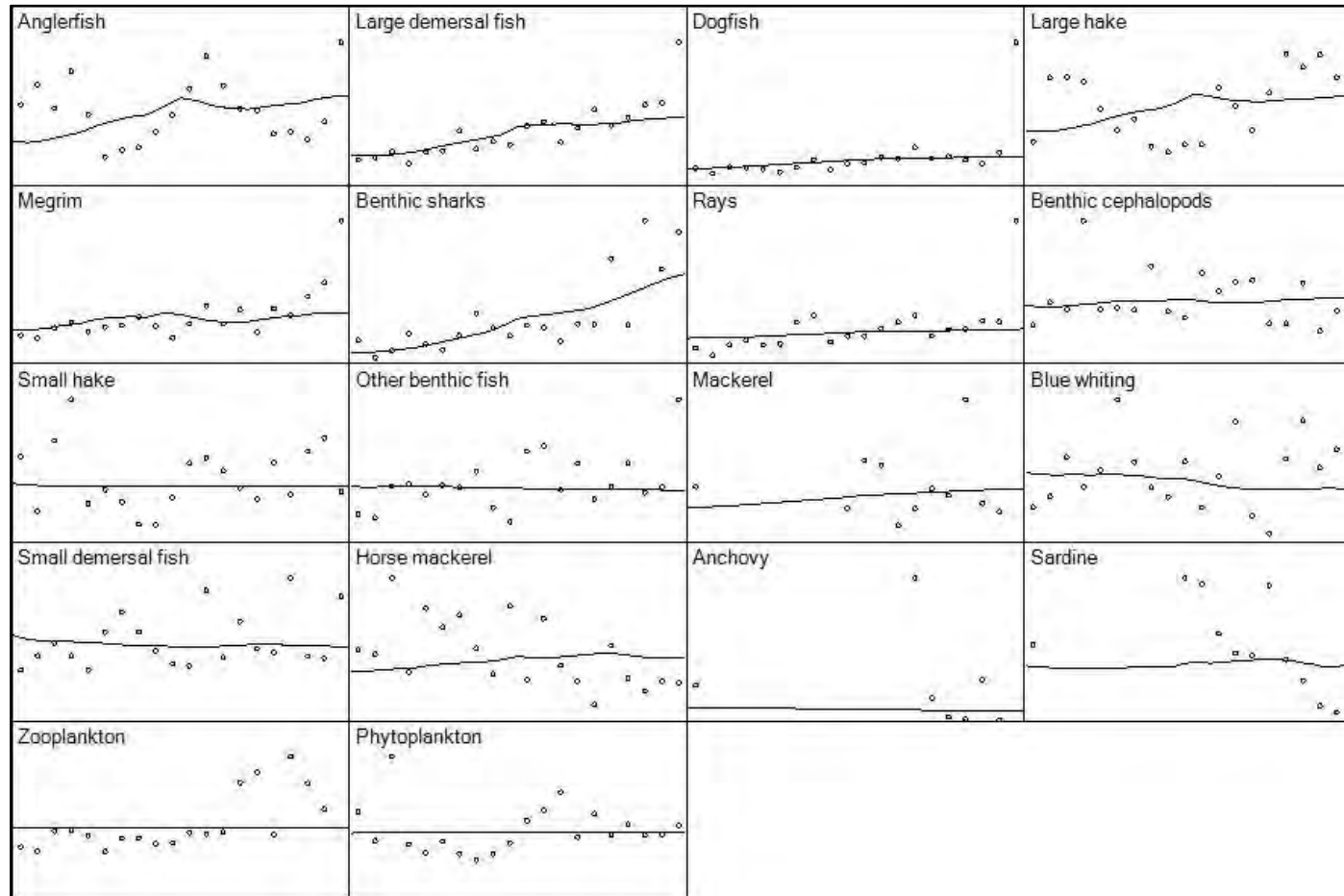
Figure 8.3. Time series biomass fitting between 1994 and 2013. The solid line represents the Ecosim fit while the dots are the observed data.

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Based on the best model output, several indicators based on the annual trophic level estimates of each functional group were computed. The mean trophic level (mTL) of the community showed different patterns depending on the threshold considered. Considering the whole community (excluding only those functional groups with TL=1, as in Shannon, 2014), the mTL showed a quite constant increase along the 20 years modeled, from 3.01 in 1994 to 3.08 in 2013 (Figure 8.4a). A similar pattern was found when the threshold was established in functional groups with TL > 4 (mTL4), with an increase in mTL4 from 4.13 to 4.22 (Figure 8.4c). Surprisingly, this trend could not be acknowledge in the mTL of trophic groups with TL>3.25 (mTL3.25), which showed a slight decrease of 0.02 units along the time series.
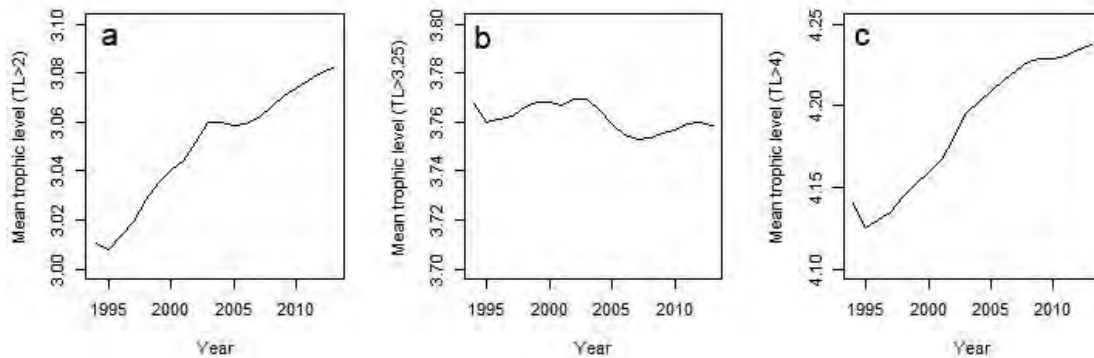
Figure 8.4. Mean trophic level of the community calculated as the weighted average of all functional groups (FG) trophic levels excluding those FG with TL = 1 (a), excluding FG with TL > 3.25 and excluding FG with TL > 4.

Considering the catch data, while a quite steep decrease in the total catch has occurred between 1994 and 2013, the mean trophic level of the catch does not show any clear trend, fluctuating around a mean value of 3.66. This can be explained by the reductions observed in all fleets in the study area, thus, the reduction of the catches of the different functional groups keep quite proportional along the time series and does not greatly affect the mean trophic level of the catch.
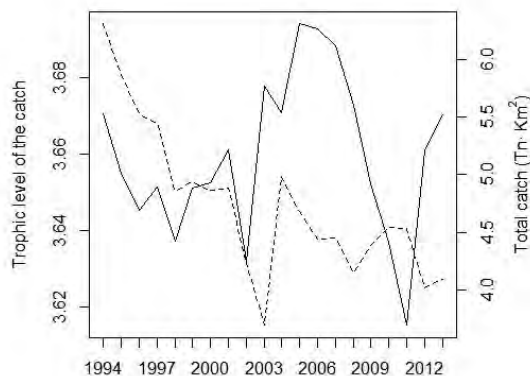
Figure 8.5. Mean trophic level of the catch (solid line) calculated as a weighted mean and total catch (dashed line) in Tn·Km2.

Socioeconomic indicators

Recent works in the area addressed socioeconomic indicators throughout models disaggregated by fleets which consider also the multiple species caught (Garcia et al., 2016, Sampedro et al., 2016). The work developed in the GADGET multispecies model has conditioned the socioeconomic indicators chosen for the EwE model since the main aim is to compare outputs for both models. There were considered two different indicators for socioeconomic model performance: revenue, measured as the value of landings and cetacean biomass as a social value of the ecosystem.

Revenue

Landings data in the area were estimated based on sale notes compiled every year by IEO until 2012. Since 2013 landings were estimated based on concurrent sampling were observed landings in the sampled vessels were extrapolated to the total fleet based on the effort measured in each metier (ICES, 2015). This data are used as input figures in the model. The Ecosim model estimates catches based on this data and the dynamic system where effort are the main driver. Species prize was taken from the Galician regional data base for fishing statistics in first sale in auctions (www.pescadegalicia.com). Galician auctions are one of the main markets for fishing landings in Spain. Fish sold in these auctions include the landings on Galician harbours but also fish coming by lorries from other countries such as Portugal, France or Ireland. To avoid that price mixing from hake coming from Northern stock (which may have different value) a selection of auctions was performed. This selection excluded the main auctions were Northern stocks are sold, such as Burela, Celeiro and Vigo.

Revenue was estimated as price (€/kg) times landings (Kg); where landings was estimated subtracting discards from total catches. Landings time series ranges from 1994 to 2013, however a consistent time series of prices are only available since 2001. The time period considered for Del 4.6 ranges between years 2001 and 2013.
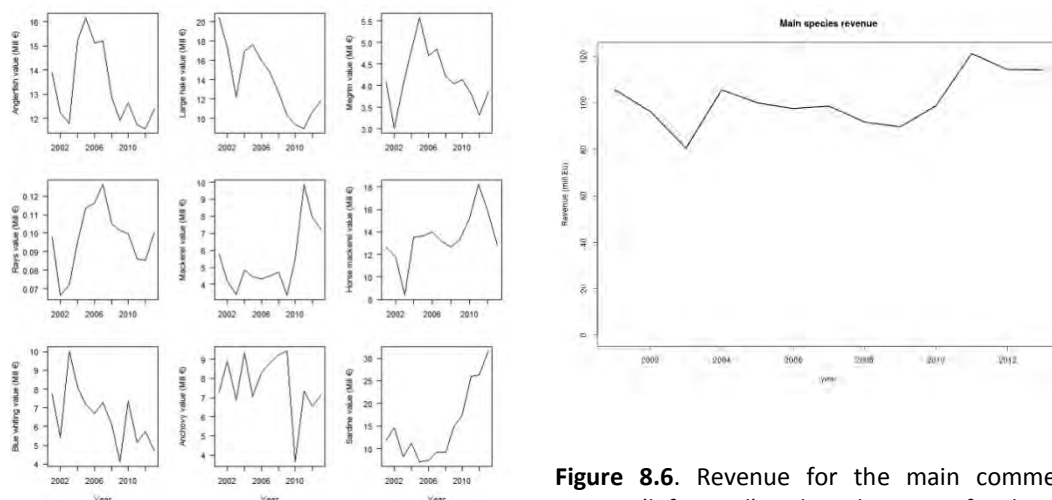
Figure 8.6. Revenue for the main commercial species (left panel) and total revenue for the same group of species (right panel) in the EwE model.

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Revenue shows quite different trends for each species (Figure 8.6). Revenue for each species along the time is quite variable with maximum and minimum ranging in twofold for most species (e.g. hake, megrim, mackerel or sardine). Yearly changes can also be quite strong for some species such as anchovy between 2009 (9 mill €) and 2010 (4 mill €) or hake between 2013 (20 mill €) and 2003 (12 mill €). There is not any species that dominates the revenue, being the sardine in recent years, with revenues near 30 mill € the bigger revenue, although it does not reach 1/4 of the total revenue (120 mill €). The total revenue is much more stable, ranging from 80 mill € in 2003 to 120 mill € in 2011 and changes in year to year revenue are also quite stable.

Dolphin biomass

It has been identified 3 ways that cetaceans provide a socioeconomic value, excluding human consumption, which is not relevant in our case of study. First there are direct activities such as on sea watching. Dolphins watching are becoming an important component on nature excursions in the area where an increasing number of companies offer sea excursions where seeing dolphins is a valuable incentive for their success. Second, people value the existence of dolphins even in the absence of opportunities for active use. There are studies that have shown that people would be willing to pay for the recovery of a cetacean population (Olar et al., 2007) or for getting a specified increase in their abundance (Larson et al., 2004). And third, dolphins are key components in the ecosystem as top predators and changes in their abundance have direct impact (positive or negative) in the health and abundance of other resources. The society believes that opportunities for the enjoyment of cetaceans are valuable and the sustainable management of cetaceans is regarded as providing a value for the future.

Dolphin biomass were developed in EwE for both dolphins together (common and bottlenose dolphin). Dolphin biomass and diet to balance the Ecopath model were estimated based on different studies developed along Mareframe project (see annex 8.1). Time series of dolphin biomass was estimated in the Ecosim dynamic model driven by fleet effort (Figure 8. 7) although trophic drivers also help to estimate dolphin biomass since prey abundance are calibrated to observed abundance in surveys, and dolphin abundance define prey consumption and, indirectly, their abundance. Dolphin biomass, as estimated by EwE, between 1994 and 2013 are quite stable without any clear trend. Biomass ranges between 343 tons in 1997 to 350 in 2005 (Fig 8.7).
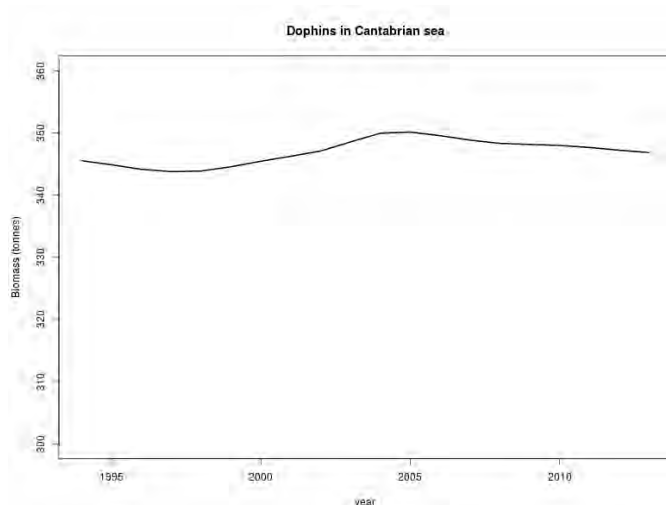
Figure 8.7. Trends in dolphin biomass (tons) in the Cantabrian Sea between 1994 and 2013.

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7.4. Discussion

In this work we develop an Ecopath with Ecosim model which serves to the MAREFRAME project for simulating the functioning of the Cantabrian Sea ecosystem. The final aim of the model would thus be aiding in the design of management strategies for the Cantabrian Sea under an ecosystem approach. As described above, the final model reproduces the biomass and catches time series of most functional groups reasonably well, and as such, we believe it can be a valuable tool to the MAREFRAME project. The fish functional groups with higher trophic levels are those whose variability and time series trends are better reproduced by the model, indicating that fishing effort is the main driver for these functional groups.

In spite of the value of the present model as a management tool, we have identified a series of shortcomings during the development of the model that need to be discussed, to be added to the limitations derived from the modeling framework. Specifically for the Cantabrian Sea Ecosystem the aggregation of species in functional groups might obscure some interesting interaction and some species could have benefited from including more multi-stanza groups, particularly those with important ontogenetic changes in their diets. Diets of most species are based on sampling carried out during the IBTS in autumn, and thus the ecosystem view is somewhat biased towards this season. Finally, to cover the lack of data we used empirical equations to calculate several input data and/or used data from neighboring ecosystem (see Annex 8.1), which lowered the quality of our data and model. Nevertheless, the model presented here is a good approximation to the Cantabrian Sea Ecosystem structure and functioning.

Based on the Ecosim output a series of ecological indicators can be calculated to assess and quantify the health status of the ecosystem and thus to determine if Good Environmental Status is achieved. Among the most common indicators, the mean trophic level of the community (MTLs) and the TLs of the landings are regularly used as ecosystem state indicators (Christensen et al., 2008). Trophic levels are also commonly used as trend indicator (Shannon et al., 2014) and have been calculated for the Cantabrian Sea in the current work. While we do not find trends in the mean trophic level of the catch, the mean trophic level of the community seems to be slowly increasing along the modeled period, possibly related with the decrease in fishing pressure that the ecosystem has experienced.

Within the Marine Strategy Framework Directive (MSFD) OSPAR is developing and standardizing some food web indicators. The mean trophic level of the community (FW4- Change in average trophic level of marine predators) is one of the common indicators are being developed and tested under the MSFD, taking into account different TL thresholds (2.0, 3.25, 4.0). Among the indicators not directly related to TLs, a large number of ecosystems can characterize the ecosystem and trends. To name few of them, the total system throughput adds all the flows through the ecosystem indicating the ecosystem size, while the ratio between total primary production and total system respiration gives an idea of ecosystem maturity (Christensen et al., 2008). Ascendancy is a measure of ecosystem growth and the development of network links (Monaco and Ulanowicz, 1997) and redundancy measures the system energy reserves indicating system's resilience (Ulanowicz, 2004, Heymans et al., 2007). The average mutual information

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measures the organization of the functional groups indicating how the flows are channeled in the ecosystem (Ulanowizc, 2004).

The ecosystem indicators used in this model are the revenue for main commercial species and the dolphin abundance. Historic trends provide the basis for economic predictions (hake revenue) based on the dynamic interactions between the fleet activity and caught species including target species and by-catch species.

The current Ecopath with Ecosim model of the Cantabrian Sea can thus serve to explore future management measures to estimate a series of ecological and socioeconomic indicators of interest to determine the state of the ecosystem state and to establish trends highly valuable to the food web criteria under the Marine Strategy Framework Directive.

7.5. Further work

Several improvements in the model could make it fit the historical data better and thus become a more valuable tool for MAREFRAME and shall be tackled in the future:

1. Compare and adjust the phytoplankton time series using Chla satellite data. The current dataset is based on monthly measures of Chla in three stations on the Cantabrian Sea continental shelf (see Annex 8.1) and could be one of the reasons behind the poor performance of forcing functions applied to the primary production of the model due to its high variance. Using monthly averaged ocean colour images, we will be able to better estimate the relative biomass of phytoplankton along the time series.

2. Use diet time series to better adjust the species vulnerabilities. For most fish functional groups annual diet estimates are available based on the sampling carried out during the Demersales IBTS survey. Using this information to adjust the vulnerabilities could improve the model performance giving a better fit to the data.

3. Test the effect of including variability of migratory species as forcing functions. This option, suggested by Christensen et al. (2008) is based on the fact that abundance of these species responds to their population dynamics which can only be understood over a much larger spatial scale than the ecosystem under study. Management strategies implemented in the Cantabrian Sea are unlikely to affect the dynamics of these species, and as such modeling the abundance variability of the migratory species through forcing functions can be justified.

4. Compare the outputs for the study area of the two proposed models within the MAREFRAME project: the EwE model and the GADGET model. *The combination of these two models, which explore the same ecosystem with a different focus and degree of detail will provide an in-depth view of how fishery and ecosystem interplay.*

5. Parameterize the data spatially with the Ecospace add-on. Having a spatial view of the ecosystem functioning, particularly in a narrow shelf ecosystem as the Cantabrian Sea characterized by its patchy habitat distribution, could aid understanding the ecosystem functioning.

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This project has received funding from
the European Union's Seventh Framework
Programme for research, technological
development and demonstration under
grant agreement no. 613571

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This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no. 613571

7.7. Supplemetnary figures and tables

Supplementary figures S1
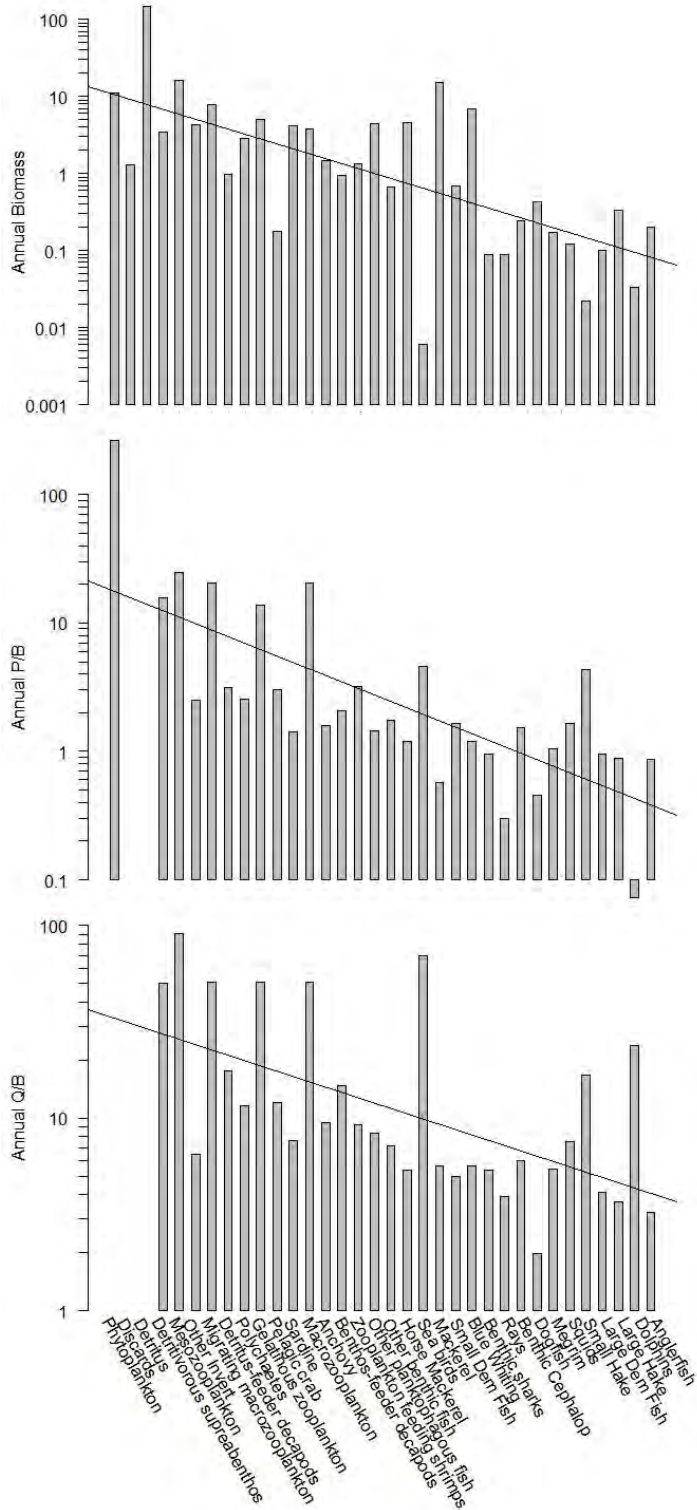
Diagnostic of the model input data using the PREBAL criteria (Link, 2010)

Prey \ predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31
1 Dolphins																															
2 Sea birds		1.00																													
3 Anglerfish				0.01																											
4 Large Dem Fish			0.35	0.01	0.13	0.01																									
5 Dogfish			0.01	0.01	0.01																										
6 Large Hake	5.17		1.05		0.57	0.52		0.01	0.01																						
7 Megrim			0.01	6.45	0.52		1.03								0.01																
8 Benthic sharks				0.01			0.01																								
9 Rays																															
10 Squids	3.10	2.00	0.01		0.01	0.01	0.01	0.01	2.83	6.85		1.16		0.01		0.22					1.69										
11 Benthic Cephalop	3.62		0.01	0.01	0.34	0.01	0.22	0.01	0.01	2.07	4.66	0.17	0.01	0.01	0.11	0.41						0.54	0.20								
12 Small Hake	3.10		1.87	0.90	2.07	2.54	1.21	0.01	3.50	1.55		4.63		0.01		0.01	0.20														
13 Other benthic fish	1.32		1.03	12.91	1.37	0.53	8.67	0.77	3.22	5.87	15.96	1.25	4.18	0.01		6.88				0.10		0.10	0.10								
14 Mackerel	0.72	2.80	0.12	1.39	0.01	9.40		4.97	0.01					0.01																	
15 Blue Whiting	43.25	9.50	57.77	21.87	20.67	49.52	0.63	9.33	3.33	21.36		14.99		3.40	0.20	1.49															
16 Small Dem Fish	9.31		6.91	7.23	1.53	3.00	2.90	0.01	3.17	8.67	22.11	8.15	0.01	0.01	0.88	1.90															
17 Horse Mackerel	10.14	6.86	26.98	0.22	5.44	11.68	23.58	23.47	0.10	1.04		8.95		0.01	6.47	0.13															
18 Anchovy	0.21	8.00		0.01	0.01	2.82	0.11	0.01	0.01	0.22		3.28		0.10	0.01	0.18	1.79														
19 Sardine	10.76	10.00	0.01	0.97	0.01	15.49		1.68	0.01	0.22				0.50		1.79															
20 Other planktophagous fish	8.28	10.00	2.93	8.40	2.11	3.64	22.43	1.44	0.98	19.66	0.54	34.97		0.20	10.16	1.36	4.51					2.57									
21 Pelagic crab		19.99	0.01	2.58	1.29			0.01	12.21	0.87			0.01			1.80						5.81									
22 Zooplankton feeding shrimps		0.47	6.19	10.97	0.68	13.99	4.13	28.92	5.37	7.77	6.82	4.21	0.01	0.80	6.80	1.20	0.50			1.00	1.99	4.10	3.85	1.80							
23 Benthos-feeder decapods				11.79	11.54	0.14	13.96	6.75	21.19	0.53	16.23	2.41	5.00	0.01	0.13	14.00				0.05	0.56		1.80		0.10						
24 Detritus-feeder decapods			0.23	5.15	17.58		5.38	0.82	2.41	0.14	11.46		12.01			16.99				0.18	2.80	0.54	3.54	2.89	0.50						
25 Polychaetes				4.90	9.37		0.25	0.53	6.85	0.95	4.22		15.61	0.13		6.93	1.79		1.58	1.00	9.01	1.72	4.06	1.70	4.52	1.12				0.10	
26 Other Invert				1.57	0.31		0.20	0.01	1.43	1.07	7.53	0.59	18.51			3.74				0.15		5.09	21.49	0.52	12.00	4.06					

#	Group	Values
27	Migrating macrozooplankton	4.00 0.09 6.15 0.01 4.43 22.83 2.27 6.84 2.82 10.29 12.91 2.52 13.40 13.27 19.55 3.60 10.00 44.53 1.69 59.74 4.80 0.99 10.00 15.00
28	Gelatinous zooplankton	0.01 0.15 0.01 0.02 0.10 0.07 7.81 2.39 3.76 11.26 1.07 2.13 2.58 1.49 5.00 5.01
29	Detritivorous supreabenthos	1.51 1.85 0.01 0.99 2.88 3.98 2.40 5.72 0.07 22.51 0.01 2.10 16.90 3.81 16.13 4.39 6.01 45.88 54.71 45.20 24.70 3.20
30	Macrozooplankton	0.01 0.01 0.01 0.01 0.50 5.95 5.00 10.23 4.40 2.35 26.18 9.99 20.00 16.94 4.13 5.68 10.15 3.53 1.12 15.00 4.62
31	Mesozooplankton	0.01 5.72 0.02 9.82 1.32 2.14 24.18 33.27 35.00 16.31 6.04 6.54 0.13 6.30 11.23 50.02 28.00 59.98 3.03
32	Phytoplankton	1.05 35.00 0.99 1.12 9.25 42.00 15.30 86.19
33	Discards	20.88 0.99 2.06 0.50 3.06 0.99 0.12 32.30 0.51 0.99 0.58 0.50 0.53
34	Detritus	0.36 13.97 5.79 1.01 31.55 27.40 56.13 37.53 100.00 10.77
35	Import	1.00 5.00 0.20 4.80 3.90 19.80 2.20 2.20 65.20 60.00 15.00 50.00 4.20

Supplementary table S1

Diet matrix composition used in the Cantabrian Sea Ecopath model (1994), with diet expressed as % of total diet biomass. Shaded cells correspond to food web links not detected in 1994, but detected other years in the diet time series. Whenever the mean interaction strength was $> 1 \cdot e^{-5}$, the link was included in the initial year with a minimum value of 0.01%.

Year	Large demersal fish	Dogfish	Large hake	Megrim	Benthic sharks	Rays	Benthic cephalo-pods	Small hake	Other benthic fish	Mackerel	Blue whiting	Small demersal fish	Horse mackerel	Anchovy	Sardine	Zoo-plankton	Phyto-plankton
1994	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
1995	1.09	0.69	2.48	0.91	0.25	0.53	1.59	0.36	0.87		1.32	1.28	0.94			0.92	0.67
1996	1.32	1.08	2.49	1.27	0.56	1.21	1.41	1.18	2.00		2.44	1.53	2.01			1.31	1.63
1997	0.84	1.03	2.39	1.46	1.30	1.49	3.69	1.66	2.07		1.58	1.28	0.69			1.32	0.63
1998	1.31	0.94	1.75	1.14	0.85	1.19	1.40	0.45	1.71		2.06	1.00	1.59			1.22	0.53
1999	1.35	0.77	1.26	1.30	0.59	1.25	1.44	0.61	2.02		4.09	1.74	1.32			0.92	0.66
2000	2.16	1.06	1.52	1.36	1.21	2.64	1.39	0.47	1.96		2.29	2.14	1.49			1.17	0.52
2001	1.43	1.52	0.89	1.65	2.14	3.06	2.52	0.21	2.53		1.57	1.75	1.02			1.17	0.45
2002	1.73	0.94	0.76	1.35	1.52	1.39	1.35	0.20	1.24		1.30	1.37	0.66			1.07	0.52
2003	1.59	1.29	0.94	0.92	1.20	1.75	1.19	0.52	0.73	0.61	2.32	1.12	1.62		1.88	1.08	0.64
2004	2.34	1.33	0.94	1.42	1.64	1.75	2.35	0.92	3.24	1.47	1.00	1.08	0.58		1.80	1.28	0.90
2005	2.51	1.68	2.25	2.05	1.53	2.24	1.88	0.98	3.44	1.38	1.88	2.57	1.44		1.15	1.25	1.02
2006	1.71	1.58	1.82	1.42	0.97	2.64	2.12	0.83	1.88	0.30	3.46	1.25	0.78		0.89	1.29	1.22
2007	2.26	2.27	1.27	1.93	1.69	3.07	2.16	0.63	2.81	0.61	0.75	1.95	0.56	4.01	0.86	2.23	0.71
2008	3.00	1.61	2.13	1.12	1.67	1.78	1.04	0.50	1.54	0.96	0.25	1.42	0.23	0.65	1.78	2.43	0.98
2009	2.37	1.71	3.02	1.96	4.48	2.16	1.04	0.93	1.99	0.84	2.39	1.35	1.06	0.11	0.81	1.24	0.74
2010	2.67	1.50	2.72	1.72	1.65	2.22	2.08	0.56	2.82	2.56	3.49	2.81	0.60	0.06	0.53	2.73	0.86
2011	3.19	1.27	3.01	2.40	6.09	2.72	0.85	1.06	1.77	0.70	2.14	1.27	0.42	1.16	0.20	2.22	0.74
2012	3.25	1.94	2.48	2.89	4.04	2.67	1.37	1.21	1.97	0.55	2.66	1.23	0.56	0.01	0.11	1.73	0.74
2013	5.65	8.54	3.29	5.08	5.61	9.08	2.38	0.59	5.09		2.00	2.45	0.54				0.84

Supplementary table S2. Relative biomasses of the functional groups used in Ecosim fitting.

7.8.   Annex 7.1

1. DOLPHINS [*Delphinus delphis*, *Tursiops truncatus*]		
B	0.0329	Based on abundance estimates for Galician and Cantabrian waters (Saavedra, C. pers. comm.)
P/B	0.072	Saavedra, C. (pers. comm)
Q/B	23.76	Saavedra, C. (pers. comm.)
Diet		Based on Saavedra, C. (pers. comm.) and slightly modified according to Santos et al., 1998, Pierce and Santos, 2000, López, 2003 and Santos et al., 2013.
EE	0.138	Estimated by the model
P/Q	0.003	Estimated by the model
Bycatch	<0.001	Saavedra, C. (pers. comm), Goetz et al., 2014

2. SEABIRDS [*Larus michaelis*, *Larus fuscus*, *Larus argenteus*, *Hydrobates pelagicus*, *Phalacrocorax aristotelis*, *Puffinus mauretanicus*, *Morus bassanus*, Alcidae]		
B	0.006	Based on population estimates in the study area from Minguez et al., 1995, Mouriño and Sierra-Abraín, 1995, Munilla, 1997, Yesou, 2003, Franco et al., 2004, Álvarez and Velando, 2007 and Munilla et al., 2011
P/B	4.6	Coll et al., 2008
Q/B	70	Coll et al., 2008
Diet		Compiled out of Munilla, 1997, Valeiras et al., 2003 and Louzao, 2015.
EE	0.156	Estimated by the model
P/Q	0.066	Estimated by the model
Bycatch	<0.001	Assumed

3. ANGLERFISH [*Lophius boudegassa*, *Lophius piscatorius*]		
B	0.2	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	0.86	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	3.22	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data) and Preciado et al., 2006
EE	0.966	Estimated by the model
P/Q	0.267	Estimated by the model
Bycatch	0.017	Estimated based on Pérez et al., 1994

4. Other LARGE DEMERSAL FISH [*Chelidonicthys lucerna, Chimaera monstrosa, Conger conger, Helicolenus dactylopterus, Phycis blennoides* (> 21 cms), *Molva macrophthalma, Scorpaena scrofa, Zeus faber*]

B	0.1	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data) and Lopez- Lopez et el., 2011
P/B	0.96	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	4.140	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data), Velasco and Olaso, 1998b, Preciado et al, 2008 and Lopez- Lopez et al., 2011
EE	0.667	Estimated by the model
P/Q	0.232	Estimated by the model
Bycatch	0.003	Estimated based on Pérez et al., 1994

5. DOGFISH [*Scyliorhinus canicula, Scyliorhinus stellaris*]

B	0.434	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	0.689	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	1.980	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data) Olaso et al., 1998 and Olaso et al 2004
EE	0.415	Estimated by the model
P/Q	0.232	Estimated by the model
Bycatch	0.022	Estimated based on Pérez et al., 1994

6. LARGE HAKE [*Merluccius merluccius* ≥ 21 cms]

B	0.489	Estimated from EwE as multistanza group (lead by Small hake)
P/B	0.880	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality
Q/B	3.68	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data) and Velasco and Olaso, 1998a, Preciado et al., 2015 and Lopez-Lopez et al., 2015.
EE	0.824	Obtained from EwE
P/Q	0.239	Obtained from EwE
Bycatch	0.004	Estimated based on Pérez et al., 1994

7. MEGRIM [*Lepidorhombus boscii, Lepidorhombus whiffiagonis*]

B	0.174	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	1.040	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	5.45	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
EE	0.841	Estimated by the model
P/Q	0.191	Estimated by the model
Bycatch	0.001	Estimated based on Pérez et al., 1994

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8. BENTHIC SHARKS [*Etmopterus spinax, Galeus atlanticus, Galeus melastomus*]		
B	0.090	Based on abundance estimates from the demersal bottom trawl surveys"DEMERSALES" in 1994 (IEO unpublished data)
P/B	0.96	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	5.370	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data), Olaso et al., 2004 and Preciado et al., 2009
EE	0.753	Estimated by the model
P/Q	0.179	Estimated by the model
Bycatch	0.010	Estimated based on Pérez et al., 1994

9. RAYS [*Leucoraja naevus, Raja clavata, Raja montagui*]		
B	0.090	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	0.300	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	3.9	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
EE	0.334	Estimated by the model
P/Q	0.077	Estimated by the model
Bycatch	0.002	Estimated based on Pérez et al., 1994

10. SQUIDS [*Ilex coindetii, Loligo vulgaris, Loligo forbesi, Todaropsis eblanae, Allotheutis sp.*]		
B	0.12	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	1.658	Obtained from Brey's model
Q/B	7.5	Sanchez and Olaso (2004)
Diet		Compiled from Pierce et al., 1994, Rasero et al., 1996, Torres, 2013, Rosas-Luis et al., 2014, Valls et al., 2015 and Puerta et al., 2015.
EE	0.781	Estimated by the model
P/Q	0.221	Estimated by the model
Bycatch	0.002	Estimated based on Pérez et al., 1994

11. BENTHIC CEPHALOPODS [*Eledone cirrhosa, Octopus vulgaris, Octopus saluttii, Sepia sp.,* Sepiolidae]		
B	0.241	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	1.54	Based on Brey's model estimate
Q/B	6	Sanchez & Olaso (2004)
Diet		Compiled from Hernandez et al., 2000, Ezzeddine, 2012, Torres, 2013, Puerta et al., 2015 and unpublished data based on stomach content analysis carried out during the demersal bottom trawl surveys "DEMERSALES"
EE	0.979	Estimated by the model
P/Q	0.257	Estimated by the model
Bycatch	0.026	Estimated based on Pérez et al., 1994

12. SMALL HAKE [*Merluccius merluccius* < 21 cms]		
B	0.1	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	4.35	Assumed
Q/B	16.75	Estimated by the model
Diet		Based on stomach content analysis carried out during the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data), Velasco and Olaso, 1998a
EE	0.959	Estimated by the model
P/Q	0.260	Estimated by the model
Bycatch	0.003	Estimated based on Pérez et al., 1994

13. Other BENTHIC FISH [*Arnoglossus imperialis, Arnoglossus laterna, Blennius ocellaris, Callionymus lyra, Callionymus maculatus, Gobiidae, Microchirus variegatus, Mullus surmuletus, Solea solea, Trigla trigla*]		
B	0.675	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	1.740	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	7.14	Based on Pauly's equation (1990)
Diet		Compiled out of Gibson and Ezzi, 1979, Torres, 2013 and unpublished data based on stomach content analysis carried out during the demersal bottom trawl surveys "DEMERSALES"
EE	0.916	Estimated by the model
P/Q	0.244	Estimated by the model
Bycatch	0.033	Estimated based on Pérez et al., 1994

14. MACKEREL [*Scomber scombrus, Scomber colias*]		
B	15.24	Based on abundance estimates from the acoustic surveys "PELACUS" (IEO unpublished data)
P/B	0.575	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	5.68	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl surveys "DEMERSALES" and the acoustic surveys "PELACUS" (IEO unpublished data) and data compiled out of Cabral and Murta 2002, Olaso et al., 2005 and Bachiller, 2012.
EE	0.183	Estimated by the model
P/Q	0.101	Estimated by the model
Bycatch	0.102	Estimated based on Pérez et al., 1994

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15. BLUE WHITING [*Micromesistius poutassou*]		
B	7.0	Based on SEFOS report (Anonimous, 1997)
P/B	1.2	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	5.61	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl surveys "DEMERSALES" and the acoustic surveys "PELACUS" (IEO unpublished data), Lopez-Lopez et al. (under review)
EE	0.946	Estimated by the model
P/Q	0.214	Estimated by the model
Bycatch	0.257	Estimated based on Pérez et al., 1994

16. SMALL DEMERSAL FISH [*Chelidonichtys cuculus, Chelidonichthys obscurus, Eutrigla gurnardus, Lepidotrigla dieuzeidei, Pagellus acarne, Pagellus bogaraveo, Pagellus erythrinus, Trachinus draco, Trisopterus luscus, Trisopterus minutus*]		
B	0.69	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	1.64	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	5.0	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the demersal bottom trawl surveys "DEMERSALES"(IEO unpublished data) and Lopez-Lopez et al., 2011
EE	0.941	Estimated by the model
P/Q	0.328	Estimated by the model
Bycatch	0.022	Estimated based on Pérez et al., 1994

17. HORSE MACKEREL [*Trachurus trachurus, Trachurus mediterraneus*]		
B	4.524	Based on abundance estimates from the acoustic surveys "PELACUS" (IEO unpublished data)
P/B	1.196	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	5.40	Based on Pauly's equation (1990)
Diet		Compiled out of Olaso et al., 1999, Cabral and Murta, 2002, Garrido and Murta, 2011 and Bachiller, 2012
EE	0.962	Estimated by the model
P/Q	0.221	Estimated by the model
Bycatch	0.396	Estimated based on Pérez et al., 1994

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18. ANCHOVY [*Engraulis encrasicolus*]		
B	1.448	Based on abundance estimates from the acoustic surveys "PELACUS" (IEO unpublished data)
P/B	1.58	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	9.51	Based on Pauly's equation (1990)
Diet		Based on stomach content analysis carried out during the acoustic surveys "PELACUS" (IEO unpublished data) and data compiled out of Plouvenez and Champalbert, 1999 and Bachiller, 2012.
EE	0.510	Estimated by the model
P/Q	0.166	Estimated by the model
Bycatch	0.001	Estimated based on Pérez et al., 1994

19. SARDINE [*Sardina pilchardus*]		
B	4.152	Based on abundance estimates from the acoustic surveys "PELACUS" (IEO unpublished data)
P/B	1.416	P/B = Z = F + M. F = Y/B. M based on Pauly's equation (1980) for natural mortality.
Q/B	7.65	Based on Pauly's equation (1990)
Diet		Compiled from Garrido et al.,2008 and Bachiller,2012.
EE	0.608	Estimated by the model
P/Q	0.185	Estimated by the model
Bycatch	0.008	Estimated based on Pérez et al., 1994

20. Other PLANKTOPHAGOUS FISH [*Argentina sphyraena, Boops boops, Capros aper, Gadiculus argenteus, Macroramphosus scolopax*]		
B	4.5	Based on abundance estimates from the acoustic surveys "PELACUS" (IEO unpublished data)
P/B	1.45	Based on Pauly's equation (1980).
Q/B	8.40	Based on Pauly's equation (1990).
Diet		Compiled out of Halliday, 1969, Lopes et al., 2006, Bachiller, 2012 and Lopez-Lopez et al. (under review).
EE	0.930	Estimated by the model
P/Q	0.173	Estimated by the model
Bycatch	0.039	Estimated based on Pérez et al., 1994

21. PELAGIC CRAB [*Polybius henslowii*]		
B	0.175	Based on abundance estimates from the demersal bottom surveys "DEMERSALES" in 1994 (IEO unpublished data)
P/B	3.011	Obtained from Brey's model (2012)
Q/B	11.99	Computed using Cammen equation (1980)
Diet		Based on Signa et al., 2008 (modified for considering net feeding and scavenging behaviour)
EE	0.945	Estimated by the model
P/Q	0.251	Estimated by the model
Bycatch	0.080	Estimated based on Pérez et al., 1994

22. ZOOPLANKTON FEEDING SHRIMPS [*Chlorotocus crassicornis, Dichelopandalus bonnieri, Pasiphaea sivado, Plesionika heterocarpus, Processa canaliculata, Solenocera membranacea*]		
B	1.35	Based on abundance estimates from the demersal trawl bottom survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	3.210	Coll et al. (2007)
Q/B	9.22	Computed using Pauly's model (1990)
Diet		Compiled out of Fanelli & Cartes, 2004 and Cartes et al., 2007.
EE	0.970	Estimated by the model
P/Q	0.348	Estimated by the model
Bycatch	0.050	Estimated based on Pérez et al., 1994

23. BENTHOS- FEEDER DECAPODS [*Alpheus glaber, Atelecyclus sp., Cancer pagurus,* Crangonidae*, Geryon trispinosus, Goneplax rhomboides, Hommarus gammarus, Liocarcinus depurator, Macropipus tuberculatus, Macropodia longipes, Maja squinado, Neprhops norvegicus, Pandalina brevirostris, Pòlycheles typhlops*]		
B	0.960	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	2.062	Computed using Brey's model (2012)
Q/B	14.65	Computed using Cammen equation (1980)
Diet		Compiled out of Cartes and Sardá,1992, Freire et al., 1996, Cartes et al.,2007, Fanelli et al.,2009 and Chartosia et al., 2010.
EE	0.973	Estimated by the model
P/Q	0.141	Estimated by the model
Bycatch	0.120	Estimated based on Pérez et al., 1994

24. DETRITUS-FEEDER DECAPODS [Paguridae, Galatheidae]		
B	0.970	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	3.154	Computed using Brey's model (2012)
Q/B	17.57	Computed using Cammen equation (1980)
Diet		Compiled out of Cartes et al., 2007.
EE	0.977	Estimated by the model
P/Q	0.180	Estimated by the model
Bycatch	0.033	Estimated based on Pérez et al., 1994

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25. POLYCHAETES [Aphroditidae, Onuphidae, Amphinomidae, Eunicidae, Arabellidae, Lumbrineridae, Sigalonidae, Serpulidae, Sipunculida and Hirudinea]

B	2.625	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	2.53	Torres et al., 2013
Q/B	11.53	Coll et al. (2007), Coll et al. (2008)
Diet		Jumars et al., 2015
EE	0.901	Estimated by the model
P/Q	0.219	Estimated by the model
Bycatch	<0.001	Estimated based on Pérez et al., 1994

26. OTHER INVERTEBRATES [*Actinauge richardi*, Anthozoa, Ascidiacea, Balanidae, Bivalvia, Brachiopoda, Briozoa, *Charonia lampax*, Cirripedia, Crinoidea, Scaphopoda, Gastropoda, *Gracilechinus acutus*, Holothuroidea, Hydrozoa, *Leptometra celtica*, Ophiuroidea, Opistobranchia, *Parastichopus sp.*, Pennatulacea, Priapulida, Asteroidea, Echinoidea, Porifera]

B	4.34	Based on abundance estimates from the demersal bottom trawl survey "DEMERSALES" in 1994 (IEO unpublished data)
P/B	2.5	Sanchez & Olaso, 2004
Q/B	6.5	Sanchez & Olaso, 2004
Diet		Based on general knowledge on the main feeding modes of the most abundant taxa.
EE	0.992	Estimated by the model
P/Q	0.385	Estimated by the model
Bycatch	0.046	Estimated based on Pérez et al., 1994

27. MIGRATING MACROZOOPLANKTON [Euphausiacea, Mysidacea]

B	6.289	Estimated by the model
P/B	20.41	Coll et al., 2006, Torres et al., 2013
Q/B	50.94	Coll et al., 2006, Torres et al., 2013
Diet		Compiled out of Kaartvedt et al.,2002, Schmidt, 2010, Cleary et al., 2012, Chouvelon et al., 2012 and Lopez-Lopez et al. (under review).
EE	0.95	Assumption
P/Q	0.403	Estimated by the model
Bycatch	<0.001	Estimated based on Pérez et al., 1994

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28. GELATINOUS ZOOPLANKTON [Chaetognates, Tunicates, Jellyfish]		
B	5.038	Estimated by the model
P/B	13.87	Coll et al., 2006 and 2008; Torres et al., 2013
Q/B	50.48	Coll et al., 2006 and 2008; Torres et al., 2013
Diet		Compiled from Silver and Brunland, 1981; Mills, 1995; Hereu, 2010, Purcell et al., 2014
EE	0.44	Torres et al., 2013
P/Q	0.275	Estimated by the model
Bycatch	0.001	Estimated based on Pérez et al., 1994

29. DETRITIVOROUS SUPRABENTHOS [Isopoda, gammarid and caprellid amphipods and cumaceans]		
B	3.446	Estimated by the model
P/B	15.62	Cartes and Sorbe, 1999; Vilas, 2005; Torres et al., 2013
Q/B	52.12	Coll et al., 2006; Torres et al.,2013
Diet		Assumed exclusive feeding on detritus
EE	0.950	Assumption
P/Q	0.300	Estimated by the model
Bycatch	0	Estimated based on Pérez et al., 1994

30. MACROZOOPLANKTON [Hyperiids, Pteropoda, Fish larvae]		
B	3.792	Estimated by the model
P/B	20.41	Coll et al., 2006 and 2008, Torres et al., 2013
Q/B	50.94	Coll et al., 2006, Torres et al., 2013
Diet		Compiled out of Silver and Brunland, 1981, Pakhomov and Perissinotto, 1996, Le Vay et al., 2001, Morote et al., 2010 and Morote et al., 2011.
EE	0.95	Assumption
P/Q	0.401	Estimated by the model
Bycatch	0.036	Estimated based on Pérez et al., 1994

31. MESO- and MICROZOOPLANKTON		
B	16.01	Based on abundance estimates from the plankton sampling program "RADIALES" in 1994 (IEO unpublished data).
P/B	25	Guenette et al., 2001, Torres et al., 2013
Q/B	90.40	Guenette et al., 2001, Torres et al., 2013
Diet		Based on general knowledge on the feeding modes of the main taxa composing the group.
EE	0.998	Estimated by the model
P/Q	0.277	Estimated by the model

32. PHYTOPLANKTON		
B	11.0	Based on abundance estimates from the plankton sampling program "RADIALES" in 1994 (IEO unpublished data).
P/B	262.8	Based on primary production measures in the study area (Bode et al., 1996)
EE	0.488	Estimated by the model

33. DISCARDS		
B	1.316	Estimated based on Pérez et al., 1994
EE	0.984	Estimated by the model

34. DETRITUS		
B	147.94	Based on modelled detritus in the study area in Somavilla, 2010.
EE	0.188	Estimated by the model

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7.8.1. References for Annex 7.1

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This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no. 613571

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8. CHATHAM RISE CASE STUDY: ATLANTIS MODEL

8.1. Introduction

The Atlantis model (Fulton et al. 2004) was used as an alternative model for the Chatham Rise case study. Atlantis is one f a number of frameworks for end-to-end ecosystem models, and was reviewed by Plagányi (2007) as one of the best operating model within a simulation testing framework, subject to sufficient data, and otherwise still a useful framework for asking "what-if" questions. Atlantis models an ecosystem from the nutrients to the socio-economic components. It is component-structured, making it convenient to develop the models in stages, further developing as new questions or data arise to explore or inform. There have been many implementations of Atlantis models (e.g. southeast Australia, California Current, north-east United States (Fulton et al. 2011), and much development since its inception in 2000 (Fulton et al. 2016, Smith et al. 2014).

Atlantis models can provide important insights for managers regarding natural resource decisions, although it is doubtful management decisions and actions will be based solely on Atlantis (Fulton et al. 2011). Atlantis is best for exploring cumulative effects, ecosystem dynamics and general management approaches. It is intended for and has fairly recently been implemented as an operating model for Management Strategy Evaluation (MSE) (Fulton et al. 2016). It consists of dynamic two-way coupling of biophysical, human users, monitoring, assessment, managements, and socioeconomic components, which are key components for adaptive management.

Atlantis models have been developed for ecosystems in Australia, North America, Europe, UK and New Zealand (http://atlantis.cmar.csiro.au/). Table 9.1 has a summary of the purposes of these models.

Table 9.1: Summary of purpose of developed Atlantis Models (http://atlantis.cmar.csiro.au/).

Purpose	Number of models
Complexity	4
Invasives	>10
Management Strategy Evaluation	7
Fisheries	>10
Nutrients	>10
Mining & energy	0
Multiple use	4
Polution	8
Climate & acidfication	>10
Catchments	>10
Indicators	9
Management	>10

In this section the Atlantis model the Chatham Rise will be described. Details of the model structure and tuning process are described. A simulated time-series of biomass of the commercial groups from the model will be compared with estimated biomass from stock assessments and survey data.

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8.2. Methods

8.2.1. Model Structure

The model area

The Atlantis model provides a spatially explicit biophysical model of the Chatham Rise region to depths of 1300 m, bounded to the west by the 400 m depth contour within the Mernoo Gap. The balanced foodweb model (described in Deliverable 4.1) is applied to the same area, but the balanced foodweb model has no spatial structure.

Previous investigations of fish communities or fish species richness that have included Chatham Rise data have produced relatively consistent results for that area. McClatchie et al. (1997) analysed species richness from depths of 80 to 898 m depth on the Chatham Rise and Campbell Plateau. They found species-richness hotspots to be concentrated on the Chatham Rise (particularly the north Rise) and reported that richness increased with depth to reach a maximum at 500 to 1000 m. Leathwick et al. (2006) investigated factors affecting fish species richness around New Zealand, and concluded that depth was the single most important environmental predictor, with highest richness occurring at depths of 900 to 1000 m, but with a broad plateau of moderately high richness between 400 and 1100 m. Richness was higher both in waters with high surface concentrations of chlorophyll *a* and in zones of mixing of water bodies of contrasting origins (e.g. the STF). Water temperature was also important, with lower richness occurring in waters that were cooler than expected given their depth. On the Chatham Rise, richness was high relative to other areas of the New Zealand EEZ, particularly on the northern slope (Leathwick et al. 2006). Tuck et al. (2009) analysed data exclusively from a series of trawl surveys of depths 200-800 m on Chatham Rise. Species communities were found to group in adjacent depth-defined strata, but with differences between depths on the northern and southern Rise, as well as some longitudinal differentiation (Tuck et al. 2009). The STF on the Chatham Rise has been shown to serve as a discontinuity for about 70% of the small mesopelagic fish species (Robertson et al. 1978). In the study most pertinent to the Atlantis model, Bull et al. (2001) investigated depths from 200 to 800 m on Chatham Rise and concluded that mean species richness peaked at 550 to 800 m on the north Rise, and was lower on the south Rise, and in depths between 200 and 550 m. They concluded that there were marked changes in fish community composition at depths of about 350 and 550 m, and that depth, latitude, and to a lesser extent longitude explained most of the variation in abundance and composition of catches. These five investigations indicated that the division of the Chatham Rise into polygons for Atlantis modelling should occur primarily based on depth categories, with the northern and southern slopes separated (owing to the different water masses and fish communities to the north and south of the STF), and with some longitudinal differentiation as well.

A large amount of data on the biology, diet, abundance and distribution of demersal fish and invertebrate species has been collected from the series of trawl surveys of depths 200–800 m on Chatham Rise in January annually from 1992 to 2014 (Livingston et al. 2002, O'Driscoll et al. 2011). The survey area was essentially stratified by depth, latitude, and longitude. It was logical, therefore, to base the Atlantis model polygon boundaries on the trawl survey strata boundaries. Consequently, the model area was divided into 23 dynamic polygons based on bottom depth bins (< 200 m, 200–400 m, 400–600 m, 600–800 m, 800–1300 m), with bins deeper than 400 m separated into northern and southern Rise groups, and with longitudinal separation (where trawl survey strata allowed) aimed at producing western, central, and eastern polygons. This dynamic polygon area is surrounded by 6 additional non-dynamic polygons which allows for the exchange of water, nutrients and biota into and out of the

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dynamic model domain. The final configuration of the dynamic and non-dynamic polygons is shown in Figure 9.1. All model polygons are further divided into water column depth layers, ranging from one layer in some near-shore polygons to five layers for the deepest polygons. Depth layers are defined in Figure 9.1. Each polygon also contains one epibenthic and one sediment layer.
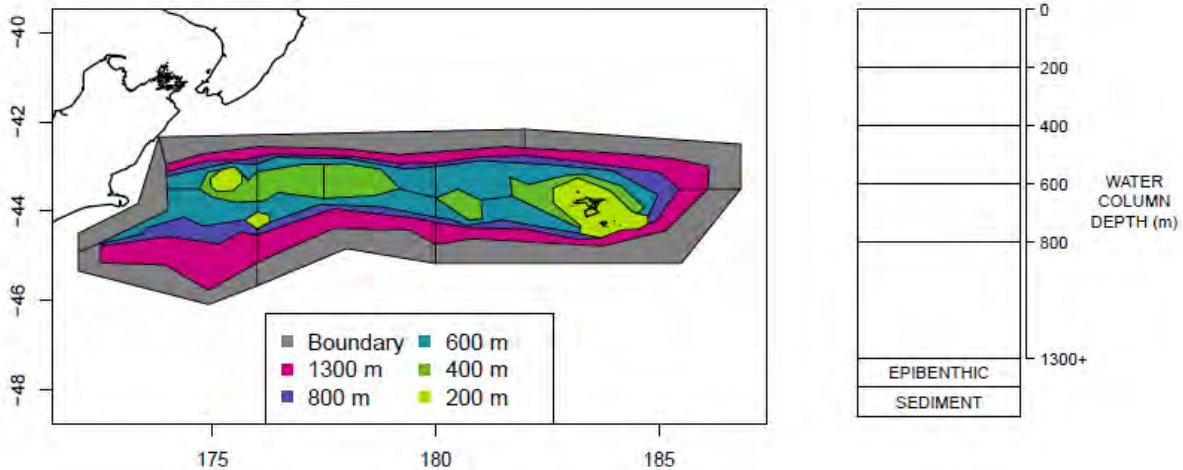
Figure 9.1. Bioregionalisation and depth structure implemented within the Chatham Rise Atlantis model.

The oceanography model

Water movement within the model region is driven by a revised Regional Ocean Modelling System model for the area (developed from Hadfield et al. 2007). This is a three-dimensional hydrodynamic ocean model of the region encompassing the Chatham Rise, forced by surface fluxes from an atmospheric reanalysis data set. The model outputs fields were validated against a number of observation-based data sets, including the CSIRO Atlas of Regional Seas 2000 (CARS 2000) climatology of sub-surface temperature and salinity, a sea surface temperature (SST) climatology from the NIWA SST Archive (NSA), and sea surface height (SSH) from the Archiving, Validation and Interpretation of Satellite data in Oceanography (AVISO) Mapped Sea Level Anomalies (MSLA) data set. The model reproduced the flow around Chatham Rise well and had a realistic seasonal cycle in the upper ocean.

The biology model

Functional groups

The model uses 53 functional groups to model the biological processes, with 15 vertebrate and one invertebrate group representing single species, and other groups representing two or more species. The main component species of the groups are presented in Table 9.2. Primary producers and most invertebrate groups are modelled as biomass pools (mg N^{-3}). Arrow squid and the general cephalopods are each divided into two biomass pools (juveniles and adults). The vertebrate groups are divided into age classes (10 age classes, with varying number of years per class, depending on the longevity of the primary species in the group), with each age class tracked by abundance (numbers) and weight-at-age (mg N). Weights are separated into structural and reserve nitrogen, with structural nitrogen (Ns) representing bones and other hard parts and reserve nitrogen (Nr) representing flesh, fat, reproductive components, and other soft tissue. The division of vertebrate groups into age classes allows for ontogenetic variations in the parameterisation.

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Table 9.2. List of functional vertebrate groups for CRAM. Name is the species group name which is the same as the species name for single-species groups. Code is a 2 or 3 letter abbreviation which each species group has in the model. Description lists the main species (in bold) and additional species in each group. Lifespan is the expected number of years an individual in that group will live failing prior mortality.

Name	Code	Description	Lifespan (years)
Baleen whales	BAL	Primarily southern right whales	80
Basketwork eel	BEE	Basketwork eels	30
Ben fish deep	BID	Benthic teleosts invertivores	20
Ben fish shal	BIS	Benthic teleosts invertivores	10
Black oreo	BOE	Black oreo	120
Bollons rattail	CBO	Bollons rattail	20
Cetacean other	CET	Primarily sperm & pilot whales, & dolphins	30
Dem fish Pisc	DPI	Demersal piscivorous teleosts	20
Epiben fish deep	EID	Epibenthic teleosts invertivores	100
Epiben fish shal	EIS	Epibenthic teleosts invertivores	10
Elasmobranch Invert	ELI	Primarily skates & dogfish	20
Elasmobranch Pisc	ELP	Primarily semi-pelagic sharks	50
Baxters dogfish	ETB	Baxters dogfish	50
Ghost shark	GSH	All chimaerids	20
Hake	HAK	Hake	30
Hoki	HOK	Hoki	20
Javelinfish	JAV	Javelinfish	10
Lookdown dory	LDO	Lookdown dory	30
Ling	LIN	Ling	30
Mackerels	MAC	Jack and blue mackerels	30
Mesopel fish Invert	MJE	Silver, white & blue warehou	20
Orange roughy	ORH	Orange roughy	120
Pelagic fish lge	PFL	Large pelagic teleosts	20
Pelagic fish med	PFM	Medium pelagic teleosts	10
Pelagic fish sml	PFS	Small pelagic teleosts	4
Pinniped	PIN	NZ fur seal	20
Reef fish	RFI	Reef teleosts	20
Seabird	SB	Seabirds & shorebirds	20
Shovelnosed dogfish	SND	Shovelnosed dogfish	40
Spiny dogfish	SPD	Spiny dogfish	30
Seaperch	SPE	Sea perch	50
Smooth oreo	SSO	Smooth oreo	100

Zooplankton

Zooplantkton are divided into four species groups, based largely on distinct species, size and diet. Table 9.3 describes the species groups, their size ranges and diet and Figure 9.2 shows their prey groups as a proportion of prey biomass. Gelatinous zooplankton (ZG) range in 1–200 mm in size and eat 30% and 40% of carnivorous zooplankton and mesozooplankton respectively and small (less than 1%) proportions of labile detritus and refractory detritus (DR). Carnivorous zooplankton are 20–200 mm, and eat a large proportion of the mesozooplankton (ZM) population and smaller proportions of the carnivorous zooplankton (ZL) and diatom (PL) biomass'. Mesozooplankton (ZM) are 0.2–20 mm and eat up to 17% of the diatom biomass and smaller proportions of labile detritus (DL) and microzooplankton (ZS). Microzooplankton, the smallest of the zooplanton groups, are 20–200 mm and eat up to 23% of picophytoplankton (PS) biomass, and small proportions (less than 1%) of diatoms and microzooplankton.

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Table 9.3. Zooplankton group definitions for CRAM.

Code	Name	Description	Size
ZG	Gelatinous zooplankton	Salps, ctenophores, jellyfish	1–200 mm
ZL	Carnivorous zooplankton	Planktonic animals e.g. pteropods, chaetognaths, natant decapods, isopods	20–200 mm
ZM	Mesozooplankton	Planktonic animals e.g. copepods, euphausiids, isopods, cladocerans, ostracods, chaetognaths, pteropods, heteropods	0.2–20 mm
ZS	Microzooplankton	Heterotrophic plankton e.g. ciliates, radiolarians, rotiferans, foraminiferans, plankton larvae	20–200mm
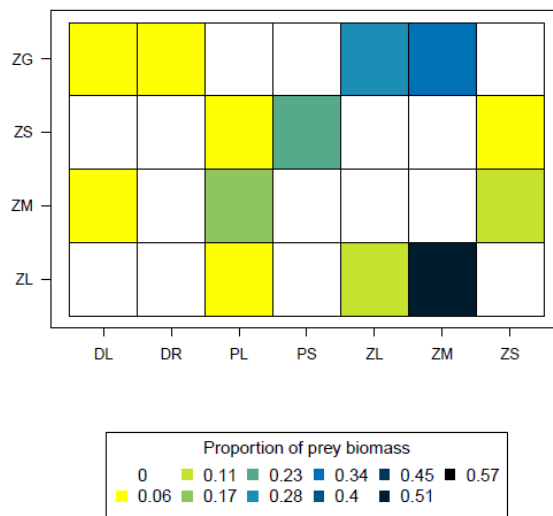
Figure 9.2. Prey species groups (x-axis) of zooplankton species (y-axis). Colour of intercepting polygon indicates the maximum proportion of prey biomass available to the zooplankton predator to eat. See Table 9.3 for zooplankton species group code descriptions and text above for prey species group code definitions.

Biological parameters

Initial biomass (B0) and initial recruitment (R0) were estimated using a Bayesian age-structured model (Bull et al. 2012). This incorporated (where available) catch history, abundance indices and life cycle parameters and estimated B_0 and R_0. B_0 was then split into age classes (cohorts as defined in Atlantis) as abundance (numbers) per age-class and mean weight (mg N) per individual in each age class. This defined the initital conditions for the CRAM. The initial abundance of mature cohorts (N_0) and R_0 together with steepness (h) were used for Beverton-Holt recruitment parameters (Figure 9.3).
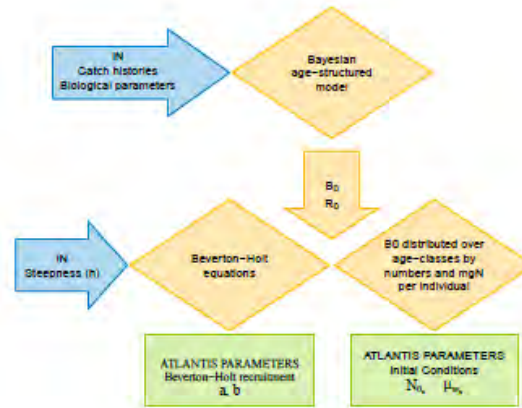
Figure 9.3. Flowchart showing the process of taking catch histories and biology parameters through to Atlantis inputs.

Natural mortality

Total natural mortality, to retain initial conditions, can be calculated on base numbers in each age-class in the initial conditions. Instantaneous natural mortality per year are as in Equation 1 for adults and Equation 2 for juveniles. Daily natural mortality as a proportion of current abundance is then as in Equation 3. If a group has no predators, linear mortality (mL; Table 9.4) should be set to the daily proportional mortality (M_d). If the group has predators, mL should be somewhere between 0 and M_d. Where in this range depends on the realised predation, and hence the best way to parameterise these can be by analysing many sensitivity runs.

$$M_{y_a} = \frac{-ln(\frac{N_C}{N_{CM}})}{AgeClassSize * NumMatureAgeClass}$$

1

where

M_{ya} is the instantaneous natural mortality (per year) NC is number of individuals in the last (oldest) age class,
N_{CM} is the number of individuals in the first (youngest) mature age class,
$AgeClassSize$ is the number of years per age class,
$NumMatureAgeClasses$ is the number of mature age classes

$$M_{y_i} = \frac{-lm(N_{CM}/(\sum_{i=1}^{CM-1} N_{Ci}+R))}{AgeClassSize * (CM-1)}$$

2

where
$M_{y\,j}$ is the instantaneous natural mortality (per year) NCM is number of individuals in the first (youngest) mature age class,
N_{Ci} is the number of individuals in age class Ci,
$AgeClassSize$ is the number of years per age class,
CM is the first (youngest) mature age class

$$M_d = 1 - e^{-M_y/365}$$

3

where
M_d is daily natural mortality as a proportion of current abundance (numbers),
M_y is the instantaneous natural mortality (per year)

Table 9.4. Total natural mortality proportion per day (mL) for juveniles and adults in age structured species groups in CRAM. Blue shading indicates predators in the model, hence at least some of their natural mortality will be predation.

Species Group	Juvenile mL	Adult mL
ASQ	0.013	0.052
BAL	0.00023	3.3e-05
BEE	0.00071	0.00034
BID	0.0012	0.00048
BIS	0.0028	0.00096
BOE	0.00019	0.00011
CBO	0.0012	0.00046
CEP	0.013	0.051
CET	0.00062	9.9e-05
CRA	0.0041	0.00026
DPI	0.0012	0.00046
EID	0.00022	0.00012
EIS	0.0023	0.00048
ELI	0.0012	0.00031
ELP	0.00067	0.00021
ETB	6e-04	0.00019
GSH	0.002	0.00082
HAK	0.00081	0.00046
HOK	0.0014	0.00066
IVH	0.0048	0.00034
IVS	0.011	0.00049
JAV	0.0027	0.00084
LDO	0.00073	0.00036
LIN	0.00071	0.00034
MAC	0.001	0.00072
MJE	0.0013	6e-04
ORH	0.00019	0.00011
PFL	0.0012	0.00048
PFM	0.0022	7e-04
PFS	0.0084	0.0022
PIN	0.00099	2e-04
RFI	0.00096	0.00033
SB	0.0019	0.00029
SND	0.00092	0.00031
SPD	0.0012	0.00047
SPE	0.00037	0.00016
SSO	0.00024	0.00015

Consumption and diet

A number of studies have examined trophic linkages on the Chatham Rise (Connell et al. 2010, Dunn et al. 2010a, Dunn et al. 2010b, Dunn et al. 2009a, Dunn et al. 2009b, Dunn et al. 2013, Dunn et al. 2010c, Horn et al. 2011, Horn & Dunn 2010, Horn et al. 2013, Horn et al. 2010, Horn et al. 2012, Rosecchi et al. 1998, Stevens 2012, Stevens & Dunn 2011, Stevens et al. 2011), and additional data has also been analysed within MareFrame. Consumers in Atlantis are modelled as biomass pools, age-structured biomass pools or age-structured groups. The age-structured groups are typically used for vertebrates, while non-vertebrates are largely modelled as biomass pools. A detailed description of

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how predation is modelled within Atlantis is provided by Audzijonyte et al. (2016). Predatory interactions are modelled in a similar way for both biomass pools and age-structured groups (although age-structured groups have additional options). Feeding interactions are modelled through biomass, which in age-structured groups are then converted to numbers to track individual mortality. In Atlantis predatory interactions are determined by:

1) Physical overlap – prey and predator must be in the same cell at the same time (determined by vertical and horizontal distribution parameters), and if prey is in the sediment the predator must be able to reach it.

2) Diet connection matrix (pPREY matrix or detailed ontogenetic diet preferences) that indicate maximum availability of prey to a predator. The actual realised consumption will depend on refuge factors, but if the value in the pPREY matrix is set to 0, no predation will occur. The prey availability matrix is illustrated in Figure 9.4.

3) Gape limitation for age-structured prey – prey that is too small or too big for the predator (either age-structured or biomass pool) will not be consumed.

4) Habitat refuge.

5) Environmental factors (temperature, salinity, pH) that can modify predator's feeding rates, prey's availability and nutritional content.

Using the standard Holling type II predator response relationship, grazing term (*Gr*), the amount of biomass of a specific prey (*prey*) consumed by a consumer *CX* is calculated as:

$$Gr_{prey} = \frac{B.C.B^*_{prey}}{1+C.HT.\sum_i B_{prey,i}} \qquad\qquad 5$$

where

$$B^*_{prey} = p_{prey,CX}.\delta_{overlap}.\delta_{habitat}.\delta_{size}.B_{prey} \qquad\qquad 6$$

is the available biomass of *prey* in an area, taking into account refuge (δ) options.

B is the feeding biomass of predator *CX* in a cell (mgN m^{-3}).
B_{prey} is the biomass of prey in that cell (mgN m^{-3}).
$p_{prey,CX}$ is the maximum availability of the *prey* to the predator *CX* defined in the pPREY matrix
C is the clearance rate of predator *CX*
HT is handling time

In biomass or age-structured biomass pool consumers (*CP*) the only variable tracked is N. Flux through a consumer biomass pool is determined by growth (G_{CP}), natural mortality (M_{CP}), predation for *i* predators ($Gr_{CP,i}$), fishing mortality (F_{CP}), and optional encystment in (Ec_{in}) and out (Ec_{out}) of the system.

$$\frac{d(CP)}{dt} = G_{CP} - M_{CP} - \sum_{i=predators} Gr_{CP,i} - F_{CP} - Ec_{out} + Ec_{in} \qquad\qquad 7$$

In age-structured consumers (*CX*) the nitrogen pool is partitioned into the reserve (*RN*) and structural nitrogen (*SN*) of an average individual of each age, and the numbers of individuals per age group are tracked.

$$\frac{d(Nm_{CX})}{dt} = Nm_{CX-1,ageup} - Nm_{CX,ageup} - M_{CX} - \sum_i^{predators} Gr_{CX,i} - F_{CX} - T_{CXout} + T_{CXin} \quad 8$$

$Nm_{CX-1,ageup}$ is recruitment or ageing into the age group
$Nm_{CX,ageup}$ is the ageing up into the next age
T_{CXout} and T_{CXin} are migration out of and into the model domain
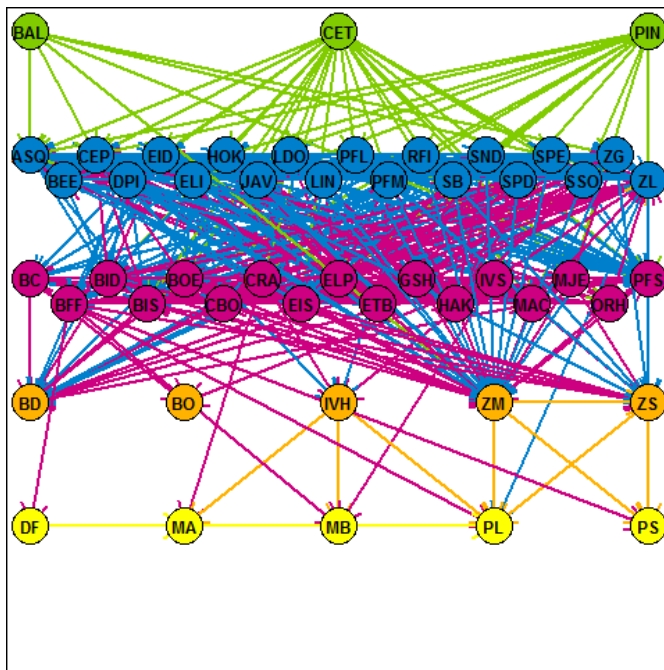
Figure 9.4. CRAM foodweb (prey availability matrix). Circle colours as follows; yellow – primary producers; orange – primary consumers; red – secondary consumers; blue – tertiary consumers; green – apex predators.

Recruitment

Recruitment of the fish groups was modelled with the Beverton-Holt function that describes the relationship between the spawning stock biomass and number of recruits. Estimates for Beverton-Holt parameters *a* and *b* were calculated using the equations below (Haddon 2011).

$$a = \frac{4hR_0}{5h-1}$$

$$b = \frac{N_0(1-h)}{5h-1}$$

where N_0 is the initial abundance in numbers, R_0 is the initial recruitment in numbers, and *h* is the steepness parameter.

The recruitment of the mammals and the seabird groups was modelled as a constant per adult individual.

MareFrame

This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no. 613571

Spatial distribution and migration

The functional groups can have different spatial distribution which can be different by season. The distribution was set as fixed for four different seasons and the model allows the distribution to be different between juveniles and adults.

For the species groups that migrate, four parameters are required for the timing of migration and the survivorship and growth while outside the model domain (Table 9.5). Those that have migrated out of the model domain return over 'Return day'≤ day < ('Return day'+'Period').

Table 9.5. Migration parameters required for species groups that migrate in and out of the model domain.

Parameter	Description
R	Return day (the day of year the group starts returning to the model domain)
P	Period (the number of days over which the group returns to the model domain)
S	Survivorship (the proportion of those that migrated to return to the model)
G	Growth (the proportional growth of individuals while outside the model domain)
L	Leave (the day of the year the group migrates out of the model domain)

The number to return on each day (N_r), is calculated from

$$N_r = S\left(\frac{N_m}{P-(t-R)}\right) \qquad\qquad 4$$

where
N_m is the number currently migrating, t is the current day of the year, and P, R and S are defined in Table 9.5. We calculated a value for survivorship based on the natural mortality parameter used for within the model domain. The migrants that return to the model domain are deemed to have grown based on a migrating growth parameter, G, and the time spent outside the model domain.

CRAM migration parameters for survivorship and growth were set to match those within the model domain, although because predator-prey interactions are not accounted for while migrating, the realised survivorship in the model domain will be less than while migrating for species groups that have predators in the model, and the growth may be less in the model domain than for those migrating if there are prey limitations in the model domain. The CRAM parameters for migrating species are in Table 9.6.

Table 9.6. CRAM migration parameters and corresponding biological parameters from within the model domain for species groups that migrate in and out of the model domain. M_L is natural mortality that is not predation while inside the CRAM model domain, L,R, P, S and G defined in Table 9.5.

Group	ML	L	R	P	S	G
Baleen whales	0.001	244	60	30	0.82	8e-04
Other cetaceans	0.001	244	60	30	0.82	0.003
Elasmobranchs - piscivores	6e-04	152	305	20	0.91	0.02
Hoki	2e-04	335	121	20	0.97	0.1
Jack mackerels	5e-04	274	1	20	0.95	0.1
Large pelagic	2e-04	152	274	20	0.97	0.2
Pinnipeds – fur seals	0.001	244	60	30	0.82	0.001
Seabirds	1e-04	91	244	30	0.98	4e-07

An example plot of the seasonal distribution of baleen whales, on the basis of Torres et al. (2013), is provided in Figure 9.5.
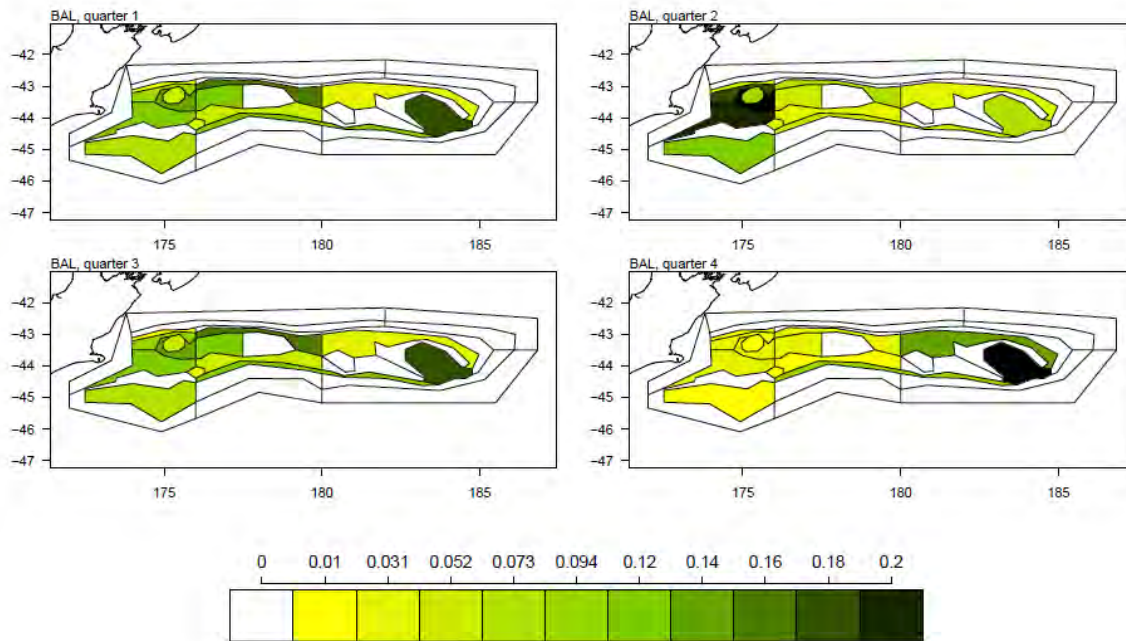
Figure 9.5. Spatial distribution for baleen whales in CRAM by quarter.

The fisheries model

The most important commercial species are harvested in the model. Each group is harvested by one fishing gear that has certain selectivity. The selectivity was chosen to be a logistic curve and size distribution of catch and survey data from the MRI was used to parameterise the selectivity curves. The harvest mortality is then multiplied with the selectivity curve which is based on length. The harvest mortality is allowed to change between years but the selectivity is the same for the whole period.

The Chatham Rise was in a near pristine state in the 1970s before commercial fishing was established (pers. comm., Matthew Dunn (NIWA)). Some commercially exploited species have been studied extensively throughout the time period over which they have been exploited. This gives us a rare opportunity to test responses of an ecosystem model to aspects of the system that feature dramatic trends and contrast. We can assess the response of a species to heavy fishing pressure in the model and compare it to studies, surveys and abundance indices. Further, we can use the model to understand how these changes may have affected other parts of the ecosystem. The Atlantis model allows forcing of historic catches, or forcing of historic effort, with catches then estimated from selectivity and available biomass. For the Chatham Rise model, we have forced catches.

Orange roughy (*Hoplostethus atlanticus)*, a species that is long living, late maturing, and has a high market value, has been heavily exploited on the Chatham Rise. Fishing began in 1979 (Francis & Clark 2005) and in the 1980's through to mid 1990's, the Chatham Rise stocks were extensively studied with a targeted trawl survey which ran every year from 1984–1990 inclusively and in years 1992, 1994 and 1995 (Clark et al. 2000). The trawl survey abundance indices declined from around 300 000 t in 1980

to around 50 000 t in 1990, which corresponded to a period of heavy exploitation, then appeared to level off. A stock reduction analysis that used data from these surveys estimated the biomass of orange roughy on the Chatham Rise to be around 20% of its unfished biomass by the mid-late 1990's. The most recent stock assessment for orange roughy on the Chatham Rise (Cordue 2014) estimated the spawning stock biomass for the northwest sub-area to have declined rapidly from the early 1980's until around 1990, then a shallower decline until the mid-2000's when it was estimated slightly below 20% of the unfished biomass (B0), then an increase up to 37% B0 in 2014 (Table 9.7). The estimated spawning stock biomass for the southern and eastern sub areas (modelled together) had a similar trajectory until 1990, when it flattened at around 25% B0 then increased slightly from 2010 to an estimated 30% B0 in 2014 (Table 9.7). The 2014 stock assessment used three main datasets: acoustic spawning biomass estimates from 1999, 2002, 2004, 2012 and 2013; commercial fishery length frequency data from 1989–2005; and trawl survey age frequency and estimate of proportion-spawning-at-age from 1994.

Table 9.7. Summary of trends in catch and spawning stock biomass for Orange roughy on the northwest (NW) and east and south (ES) Chatham Rise.

NW	Timeframe	Catch	SSB
	early 1980s	high	rapid decline
	mid 1980s–early 2000s	medium	shallower decline
	2000s	declining	to below 20% B0
	2011–2014	almost non-existent	increase up to 37% B0

SE	Timeframe	Catch	SSB
	1980s–early 1990s	high	rapid decline
	mid–late 1990s	low	flat
	mid 2000s	little higher	at 25% B0
	2011–2014	very low	increase to 30% B0

Hoki (*Macruronus novaezelandiae*) is a dominant species of the demersal fish community on the Chatham Rise (Horn & Dunn 2010) and is New Zealand's most productive finfish fishery (Horn, 2011). They mature at around 3–5 years and have a maximum lifespan of 20–25 years (Ministry for Primary Industries 2014). The trawl survey abundance estimates show hoki declined from the late 1990's until the mid 2000's when the biomass then increased. The most recent stock assessment (McKenzie 2016), which used biomass indices from trawl and acoustic surveys, proportions-at-age and sex, and proportion spawning, also showed the spawning stock biomass declining from late 1990's through to mid 2000s then increasing. The fishery developed in the early 1970's, but catches on the Chatham Rise stock were light in the 1970's, 1980's and early 1990's, then very high from mid–late 1990's through to mid-2000's, then at a constant level or around 60 000 tonnes per year from 2005 on (McKenzie 2016).

8.2.2. Indicators

Good environmental status indicators
GES indicators were calculated from the simulated model output. These indicators were: Indicator of stable biomass, trend of biomass and landings, harvest rate (HR), Shannon diversity index (SDI) and mean trophic level (MTL). The calculation of these indicators will be described in this section.

Stable biomass
A smoother was applied to the biomass trajectory using R function *lowess()* with smoother span *f*=0.3. The distance of each point in the biomass trajectory from the smoothed line was calculated. If all distances as proportions of the smoothed biomass were less than 0.2 the biomass was returned as 'stable'.

Biomass and landings trends
The median of the slope at each point divided by the median value of the trajectory (biomass or landings) was calculated. If it was less than or equal to -0.01 the slope was returned as 'decreasing', if greater than or equal to 0.01 the slope was returned as 'increasing' and otherwise returned as 'no trend'. The trend was calculated over the last 10 years.

$$\delta_t = x_t - x_{t-1}$$

where
δ_t is the slope of the trajectory at time t, x_t and x_{t-1} are the trajectory values at times t and $t-1$ respectively

Harvest rate
HR was calculated for the most important commercial groups: cod, haddock, saithe, Greenland halibut, redfish, herring, capelin and blue whiting. It was calculated as the total catch over total biomass (C/B) for each year of the simulation.

Shannon diversity index
The SDI of landings was calculated as follows:

$$SDI = \sum_{g=1}^{G} P_g \log_2 P_g$$

where P_g is the proportion of group g in total landings. SDI was calculated for each year of the simulation and included the eight most important commercial groups mentioned above.

Mean trophic level
Trophic level is assigned to each species group for adults and juveniles based on the prey availability matrix. The MTL for time *t* is calculated as follows:

$$MTL_t = \frac{\sum TL_{it} * B_{it}}{\sum B_{it}}$$

where TL is the trophic level for group *i* and B is the total biomass of group *i* at time *t*.

Economic indicators

Economic indicators were calculated for the demersal fleet targeting hoki, using a simple approach adapted from Hoskuldur and Hjorleifsson (2015), based only on recent estimates of total annual running costs and catch value. On the basis of these data, the revenue, cost and profit were calculated. It was assumed that the total vessel cost is $NZ 4.15 per kg of combined yield of hoki, hake and ling on the Chatham Rise in the year 2014, and has inverse relationship with biomass, so that the cost per kg in other years can be estimated (Eq. 1).

$$C_i = 4.15 * \frac{B_{2014}}{B_i} \qquad (1)$$

The total cost of fishing in a particular year (TC_i) is estimated from multiplying annual yield by annual cost per kg.

The price of fish was taken from Statistics New Zealand data. Revenue was calculated by estimating processed (fillet) weight from landed weight using standard conversion factors multiplied by price. Recvenue was only calculated from the yields of the three main target species, hoki, hake and ling.While these do not avccount for the full revenue of the fleet, they provide the main component, and trends observed will reflect the overall trends in revenue for the fishery. The profit then becomes Revenue – Cost.

Social indicators

Salaries to fishermen are used as a social indicator. Fishermen salaries within the hoki targeting fleet are estimated to be about 30% of the vessel running costs, and are estimated as such here.

8.3. Results and discussion

8.3.1. Sensitivity runs

Ecosystem models like Atlantis are complex, and can be very sensitive to certain parameters, over which we may be very uncertain as to the actual values. In order to determine appropriate values, the models are "tuned" to generate relatively stable ecosystem, in the absence of external drivers (e.g., environmental change or fisheries removals). An example plot of four sensitivity runs examining recruitment parameters for seabirds, baleen whales and pinnipeds (Figure 9.7) shows how abundance changes over time in relation to recruitment level, to inform the appropriate parameters to achieve a stable ecosystem. In this case, "optimal" parameters for seabirds were chosen between those of run 2 and run 3. Similar sensitivities were run for a range of key species and groups.
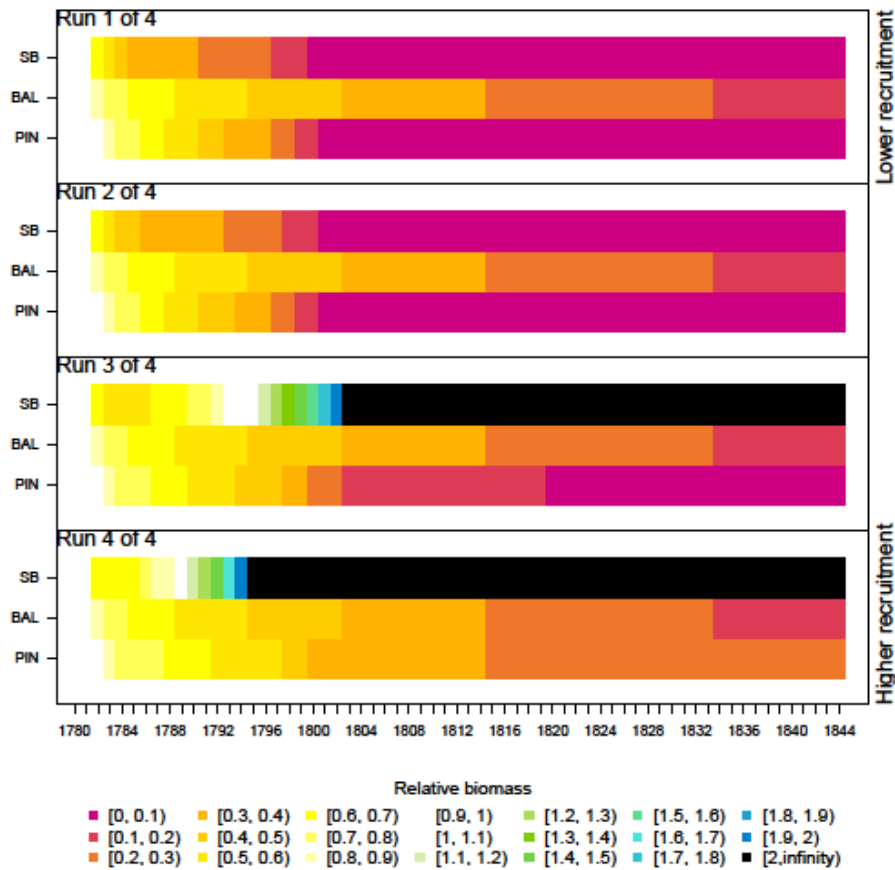
Figure 9.7. Sensitivity plots comparing seabird (SB), baleen whale (BAL) and pinniped (PIN) abundance during the model burn in period. Colour indicates abundance relative to initial abundance.

8.3.2. Primary production

Broad scale levels and patterns of primary productivity estimated by the Chatham Rise Atlantis model (Figure 9.8) were compared with satellite image derived estimates (Figure 9.9), and found to be similar.
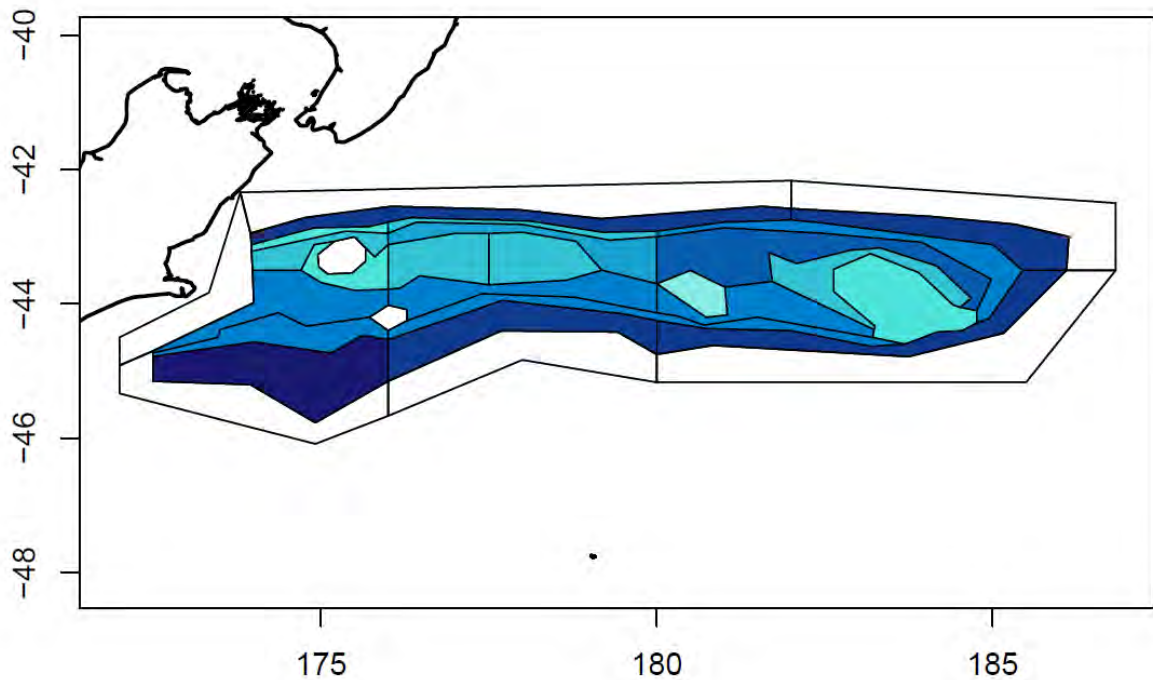
Figure 9.8. Estimated Chlorophyll a for the CRAM.
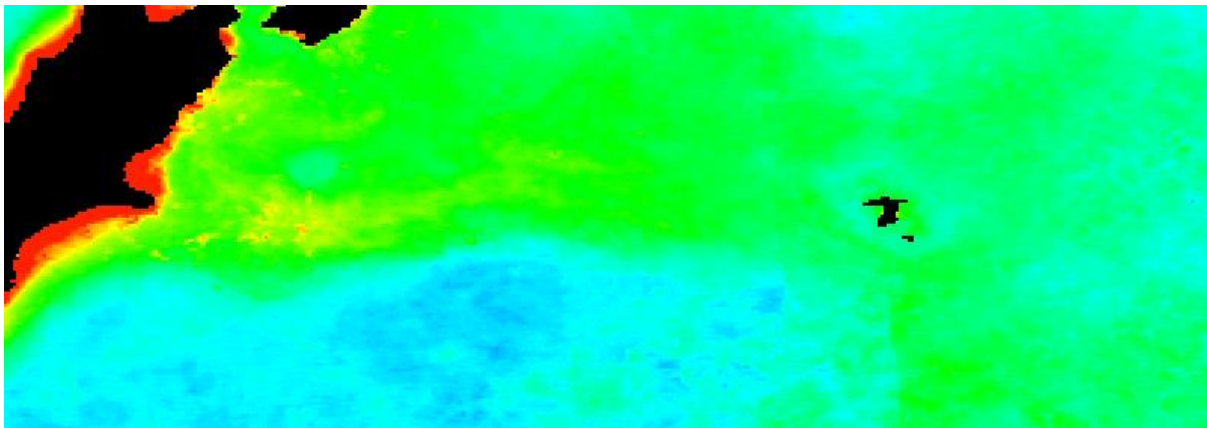
Figure 9.9. Chlorophyll a (for May, averaged from 2002 - present) from MODIS-Aqua satellite images.

8.3.3. Simulated biomass

Simulated fish stock biomass from the model was compared to biomass estimates from the Chatham
Rise trawl survey time series (O'Driscoll et al. 2011) of which we have these for 28 species groups
(Figure 9.10). The results presented here are for the un-calibrated model (the model has not been
calibrated in an attempt to match historical trends). In that respect, they are forecasts rather than
hindcasts. Most species appear to be at about the right level prior to fishing, but while some species

seem to track observed abundance changes in relation to recorded fishing removals, others rapidly collapse, suggesting further calibration is required.

MareFrame

This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no. 613571
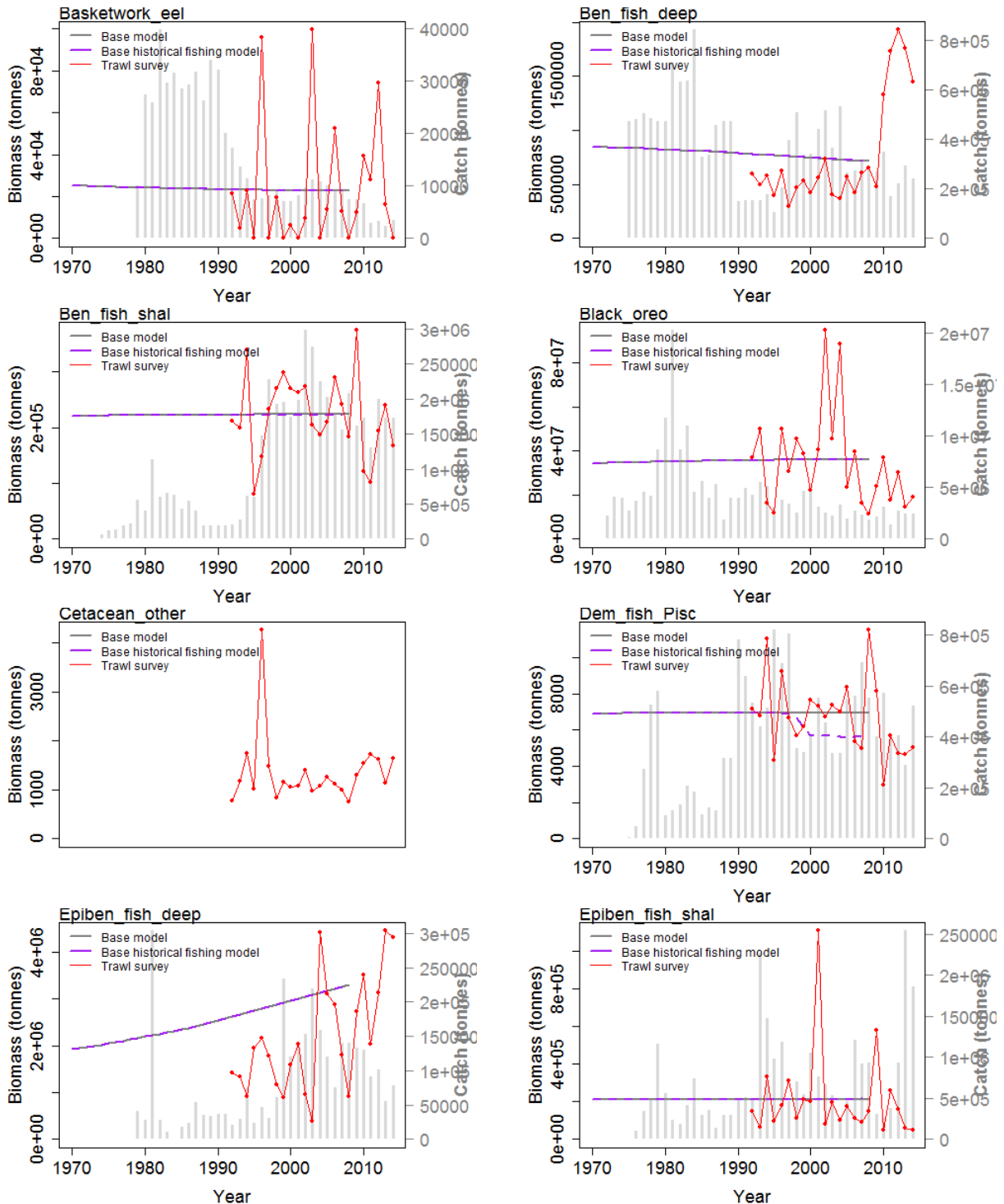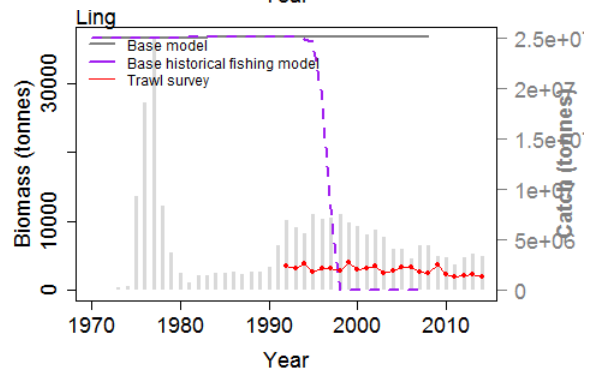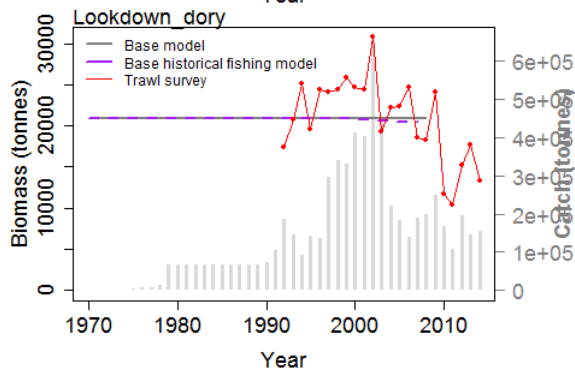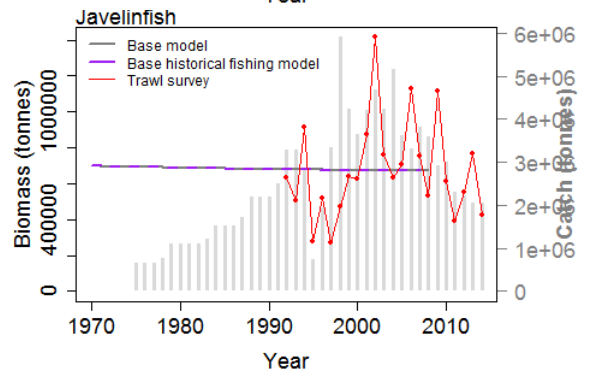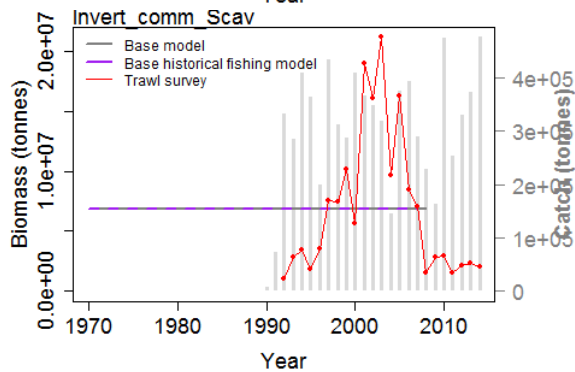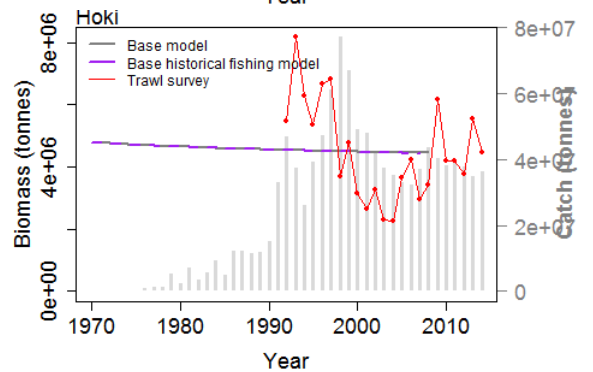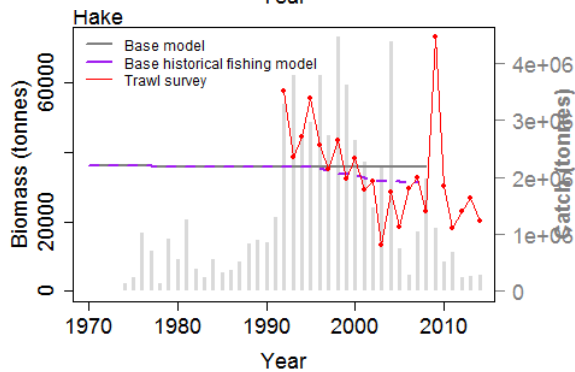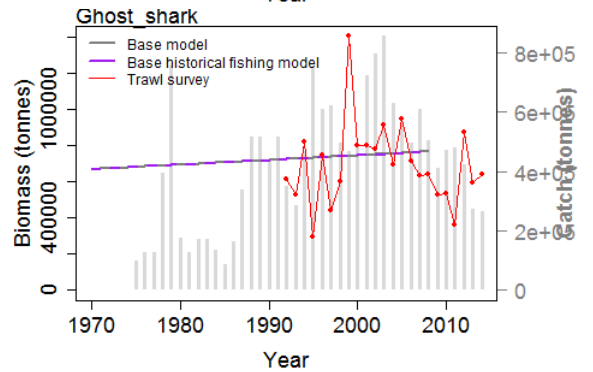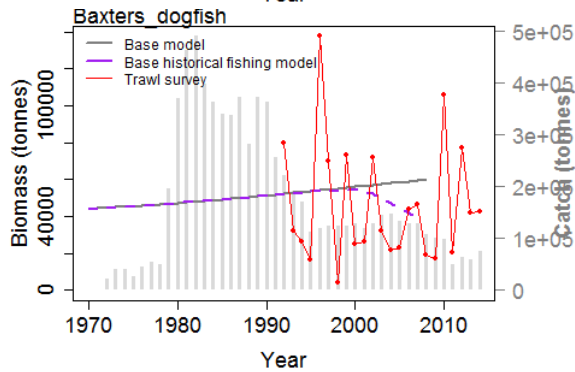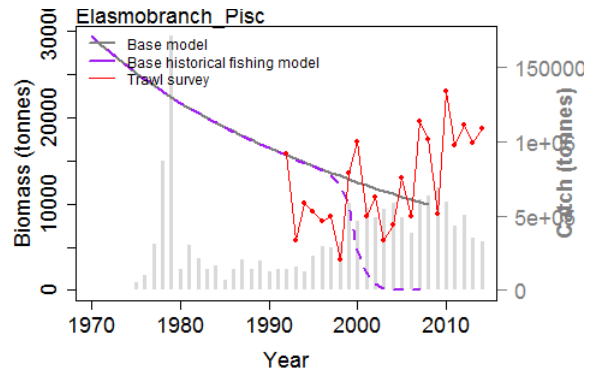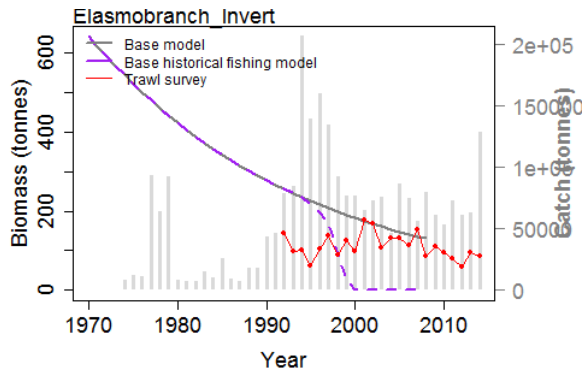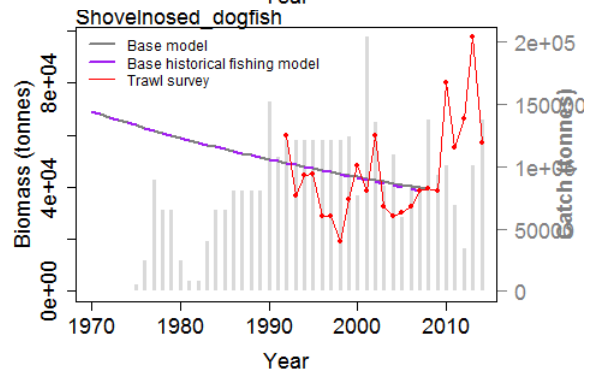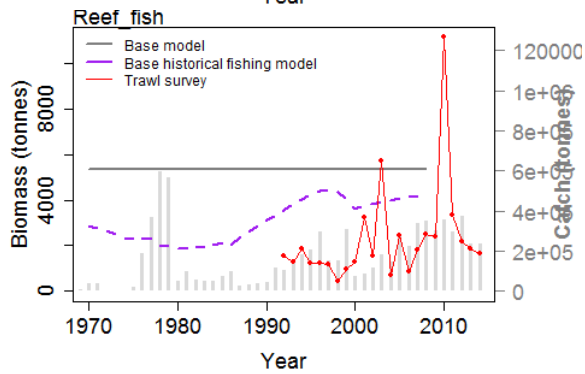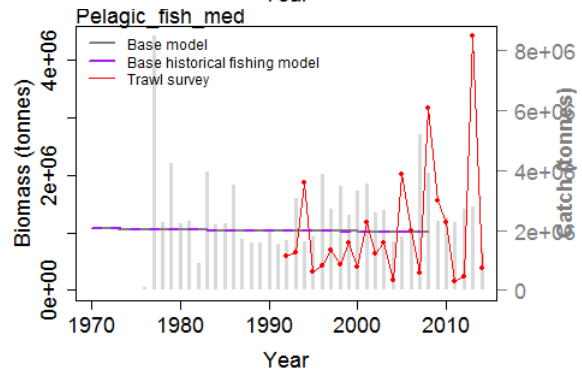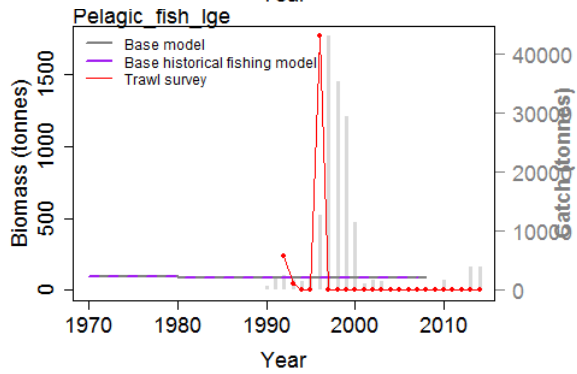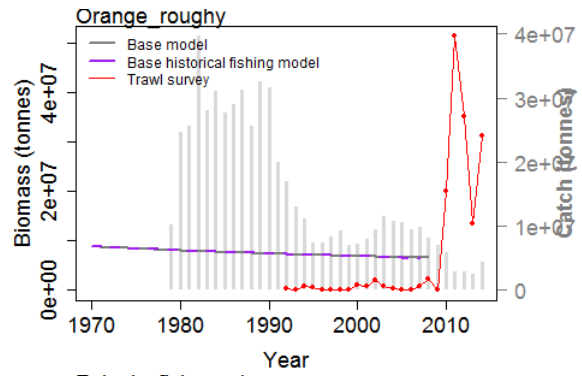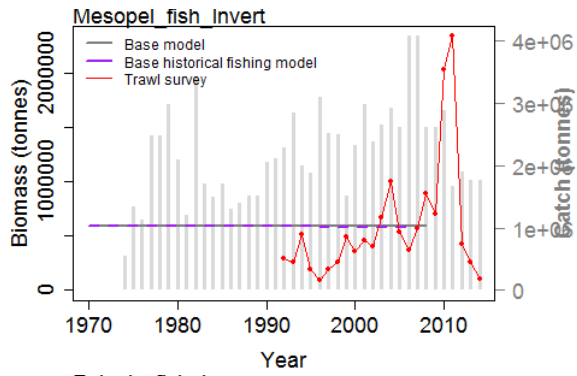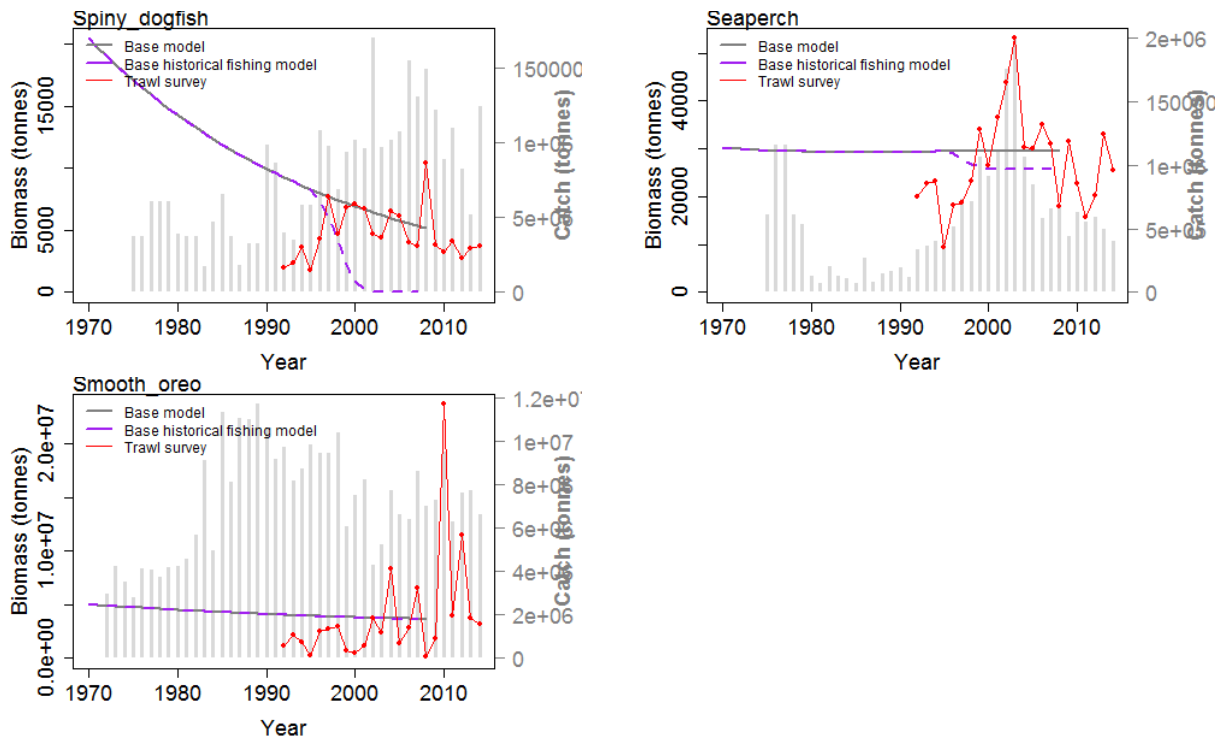
Figure 9.10. CRAM model estimates of biomass for two model runs: 1.) Base model with no fishing pressure (grey solid line); 2.) Base model with historical catches removed (purple dashed line). Biomass indices from trawl surveys are in red. Historical catches which are forced in the model (removed from the population) are shown as grey bars. These figures are for species groups for which biomass estimates from the trawl surveys exist. Cetacean-other has no forced historical catches.

The timing of the trawl survey matches well to the CRAM outputs, as we have the model outputting at the beginning of each calendar year (1st January) and the trawl surveys are carried out in January.

The trawl survey does not produce biomass estimates for all species groups – some are outside the depth range, some are outside the size range, and some are insufficiently sampled for other reasons. Abundance indices for the 27 species groups that do not have biomass estimates from the trawl survey are in Figure 9.11.
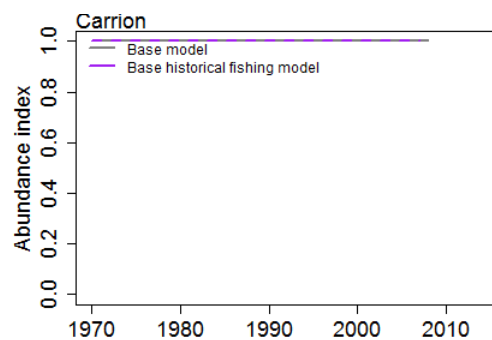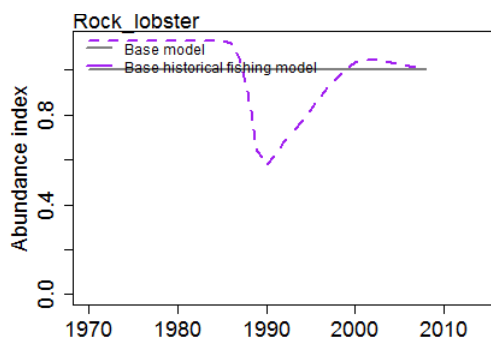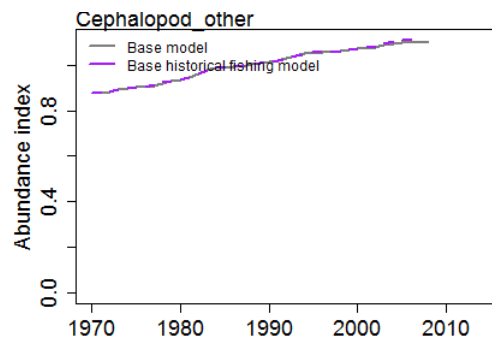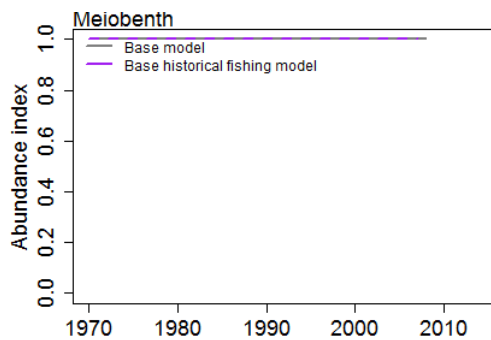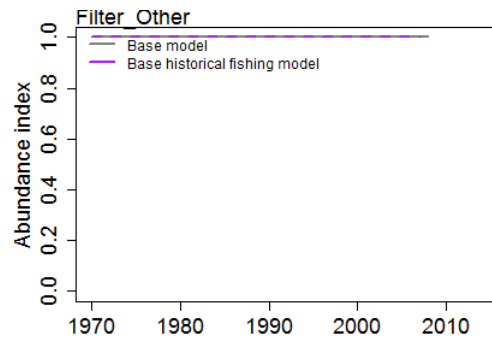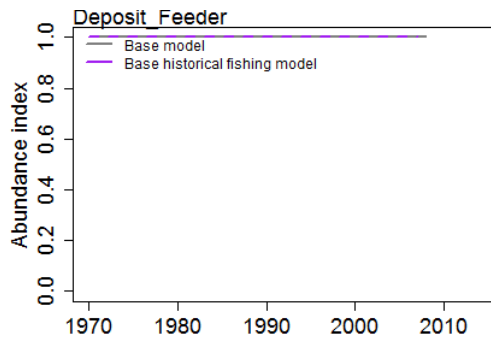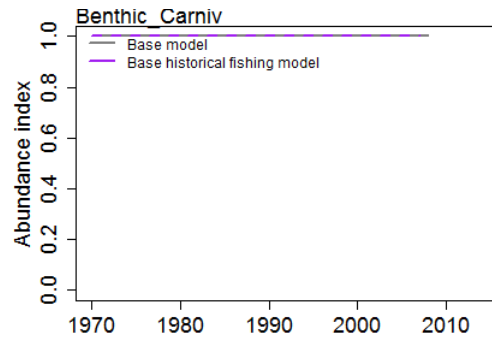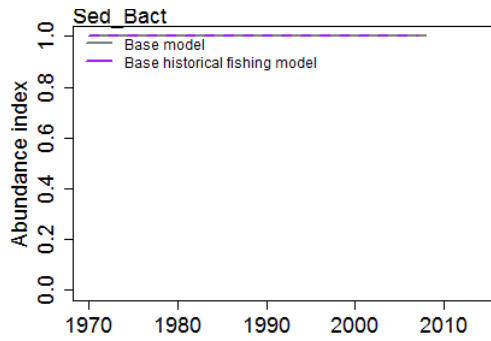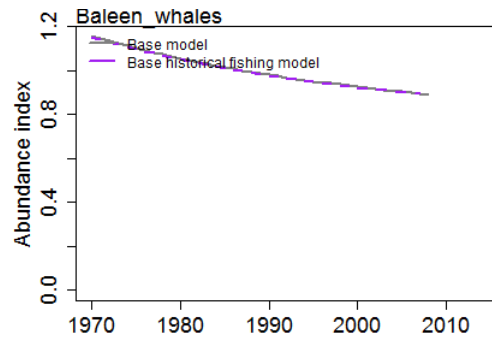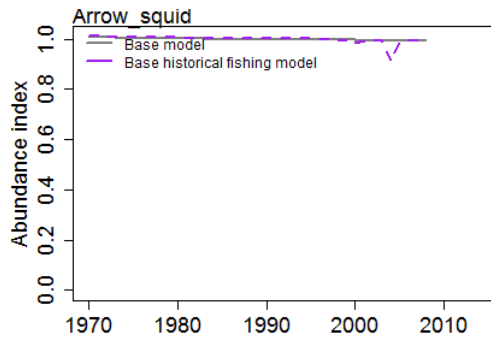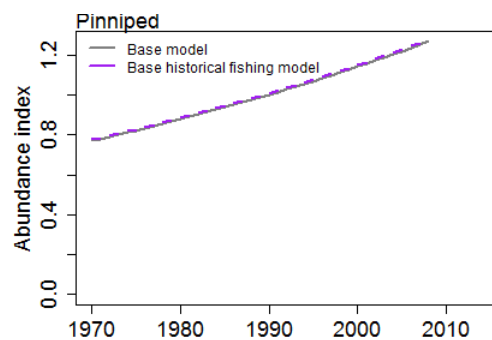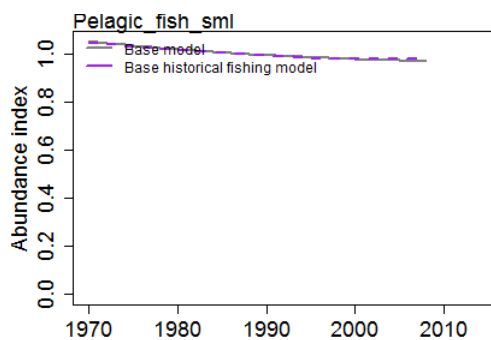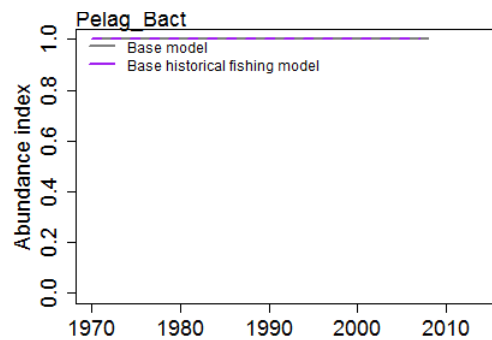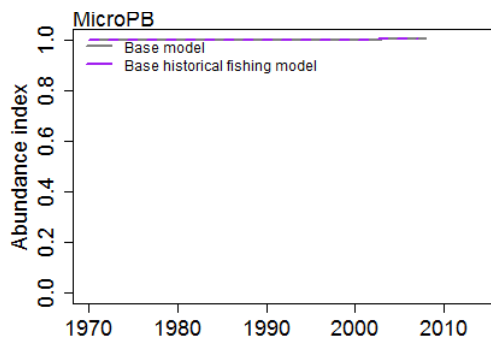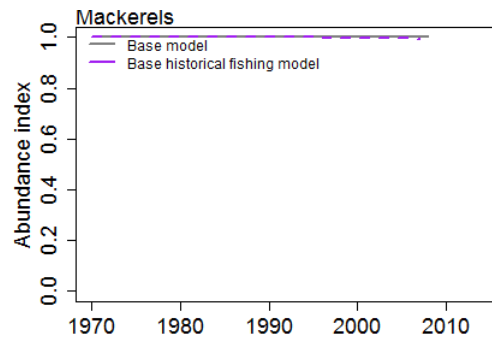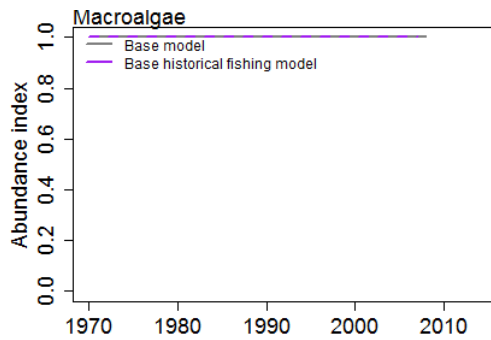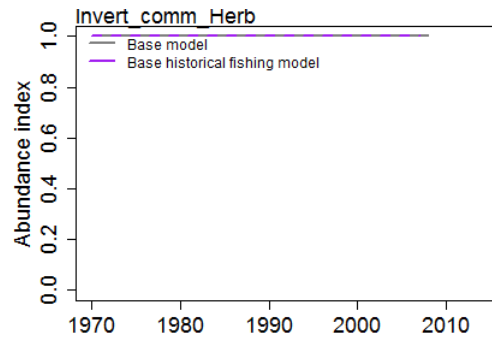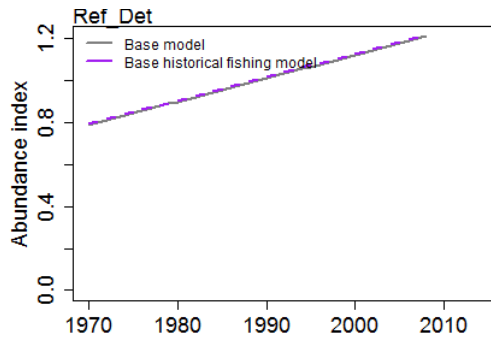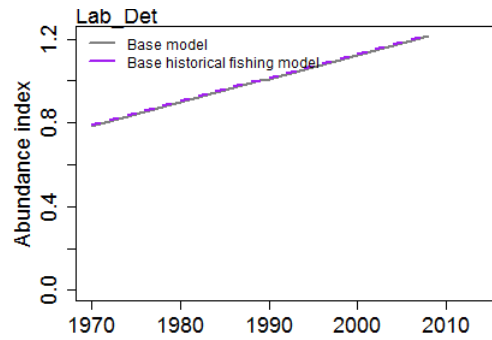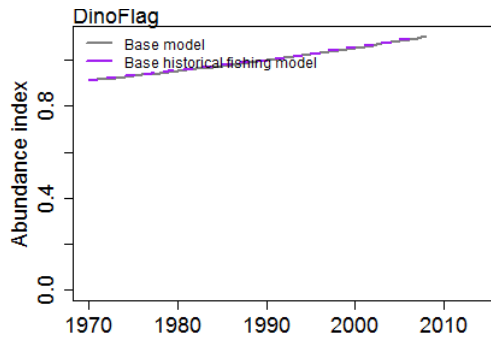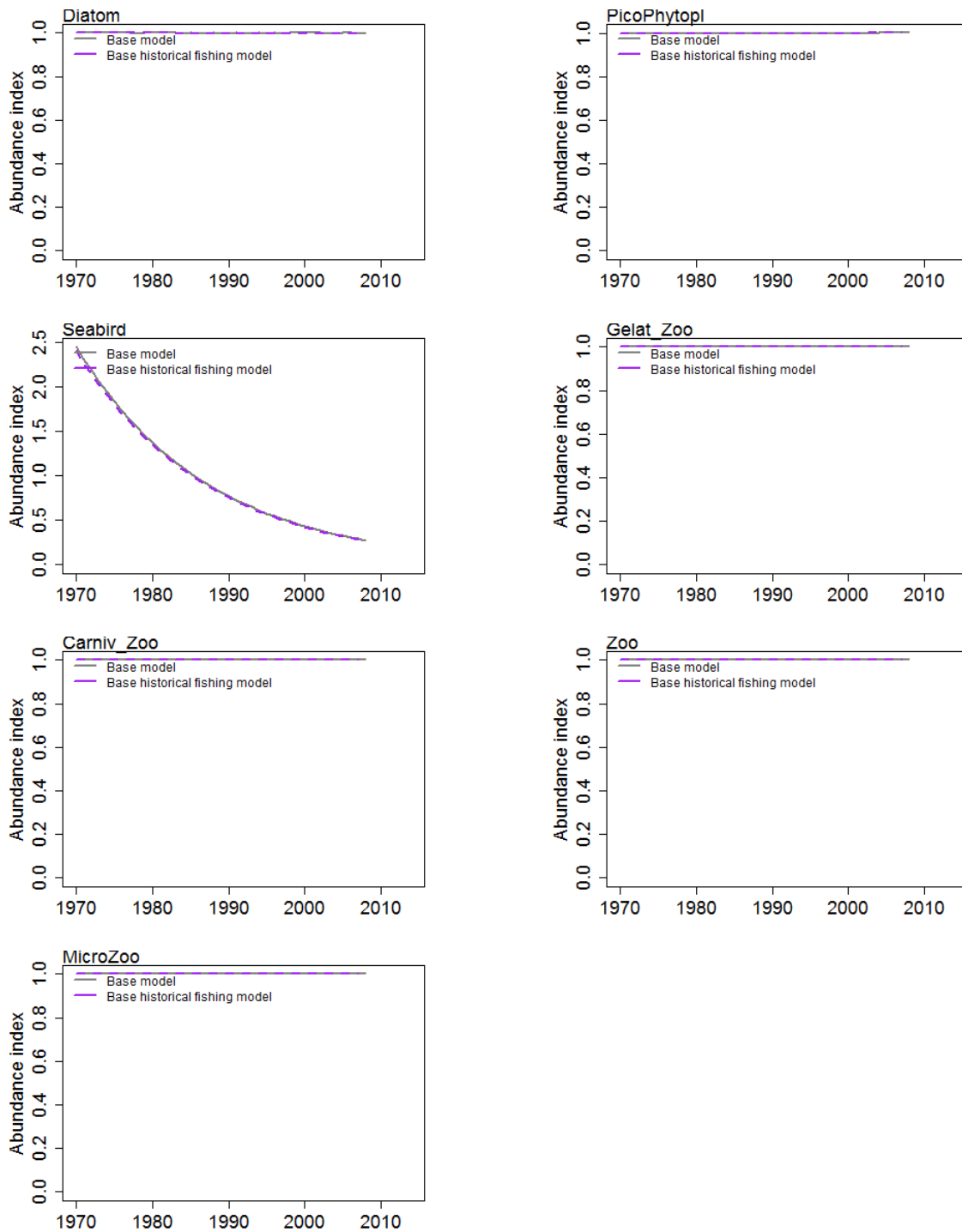
Figure 9.11. CRAM model estimates of abundance indices for two model runs: 1.) Base model with no fishing pressure (grey solid line); 2.) Base model with historical catches removed (purple dashed line). These figures are for species groups for which biomass estimates from the trawl surveys do not exist.

Good environmental status indicators

The GES indicators where calculated from the simulated output from the model. Indicators for stable biomass and trends in biomass and landings are summarised in Table 9.8. Some of the 'flat' biomass trends are the result of the species group already becoming over or under abundant then stabilising at that point. One species group was increasing, and that was EID (epibenthic fish deep).

Table 9.8. GES indicators of stable biomass and trends in biomass and landings for 34 commercial groups. These are the groups that have fishing in the model.

| Group	Stable biomass	Biomass trend	Landings trend
ASQ	TRUE	Flat	Decreasing
BEE	TRUE	Flat	Flat
BID	TRUE	Flat	Increasing
BIS	TRUE	Flat	Decreasing
BOE	TRUE	Flat	Decreasing
CBO	TRUE	Flat	Decreasing
CEP	TRUE	Flat	Flat
CRA	TRUE	Flat	Flat
DPI	TRUE	Flat	Flat
EID	TRUE	Flat	Flat
EIS	TRUE	Flat	Decreasing
ELI	TRUE	Decreasing	Decreasing
ELP	TRUE	Decreasing	Increasing
ETB	TRUE	Flat	Flat
GSH	TRUE	Flat	Decreasing
HAK	TRUE	Flat	Decreasing
HOK	TRUE	Flat	Decreasing
IVH	TRUE	Flat	Decreasing
IVS	TRUE	Flat	Decreasing
JAV	TRUE	Flat	Decreasing
LDO	TRUE	Flat	Decreasing
LIN	TRUE	Flat	Decreasing
MAC	TRUE	Flat	Decreasing
MJE	TRUE	Flat	Flat
ORH	TRUE	Decreasing	Flat
PFL	TRUE	Flat	Decreasing
PFM	TRUE	Flat	Increasing
PFS	TRUE	Flat	Decreasing
RFI	TRUE	Flat	Increasing
SND	TRUE	Decreasing	Flat
SPD	TRUE	Decreasing	Increasing
SPE	TRUE	Decreasing	Flat
SSO	TRUE	Decreasing	Flat
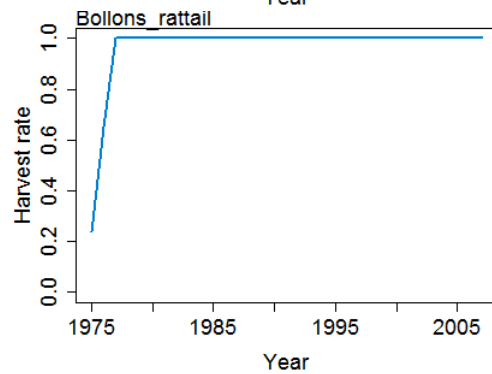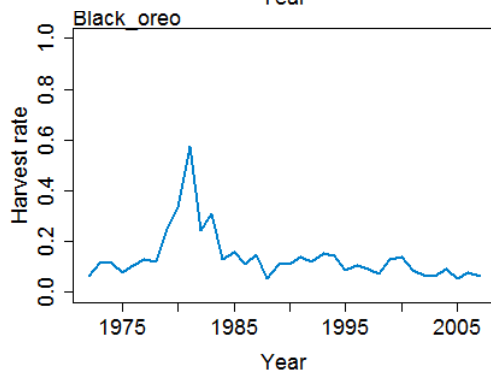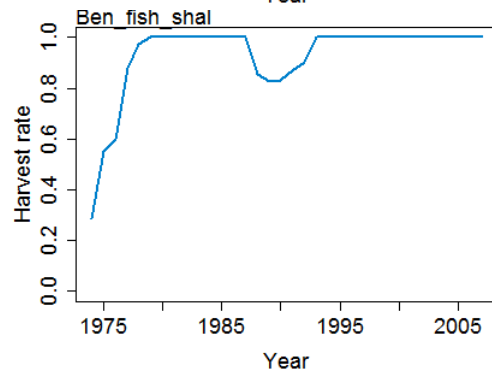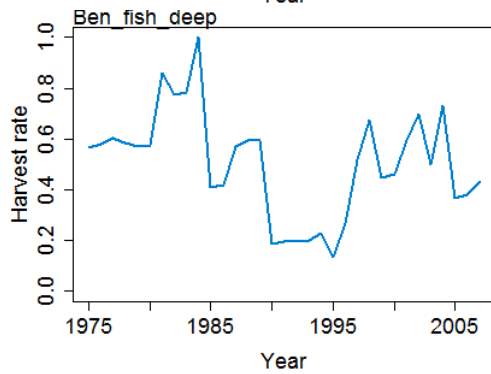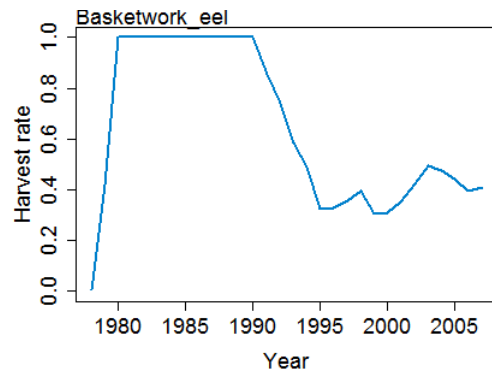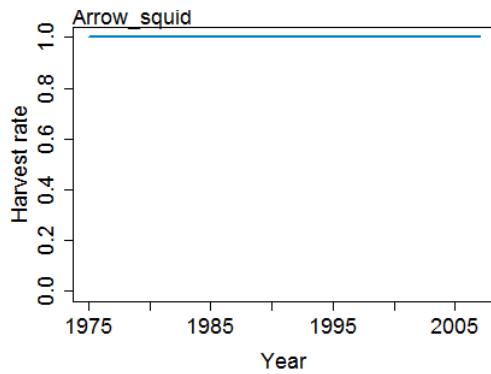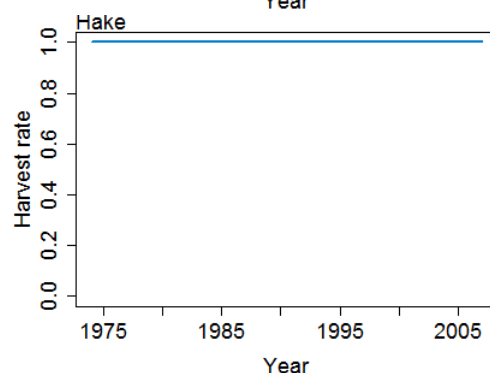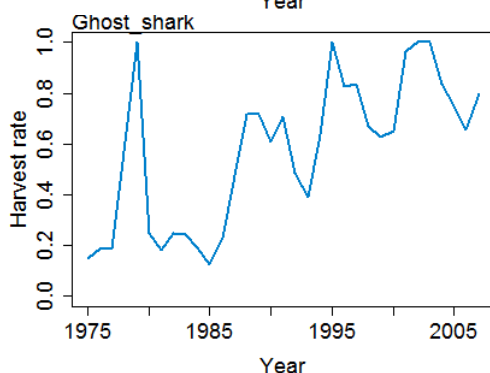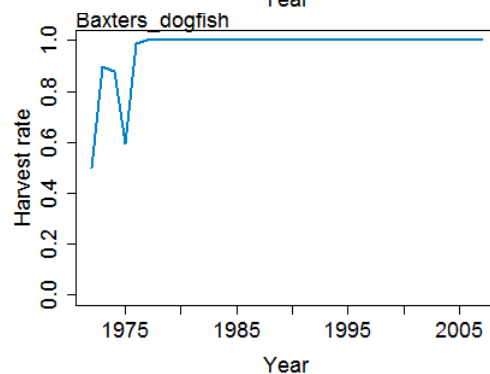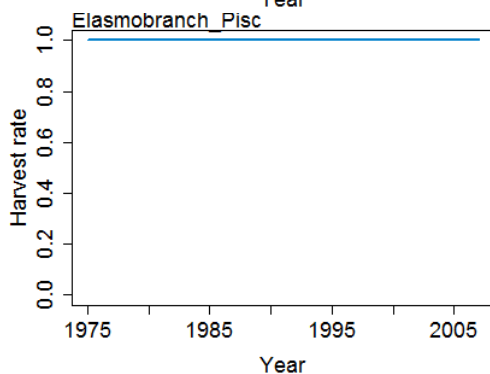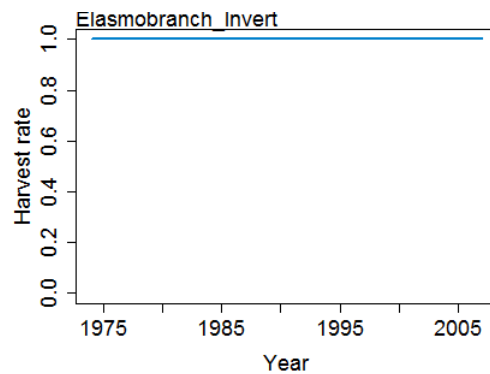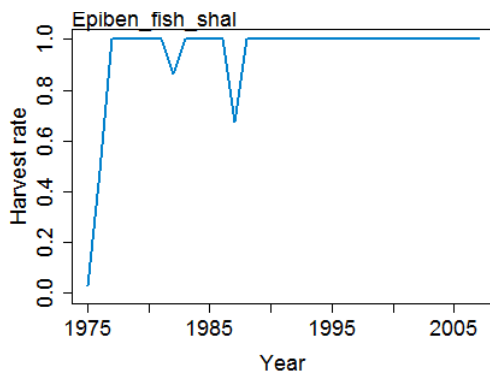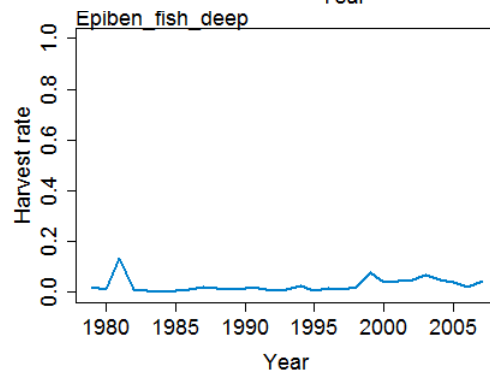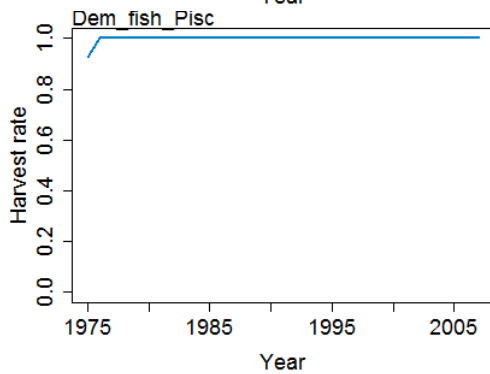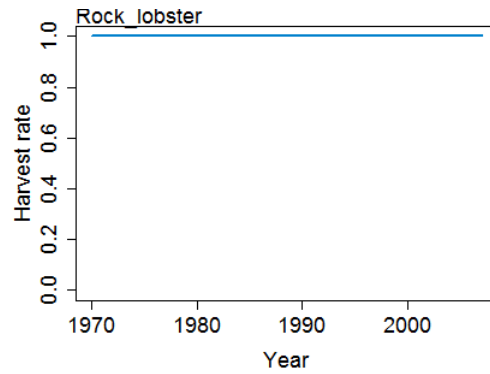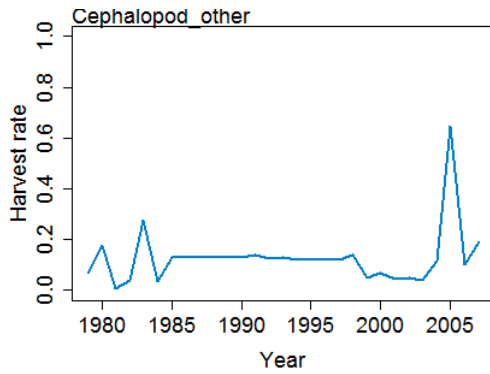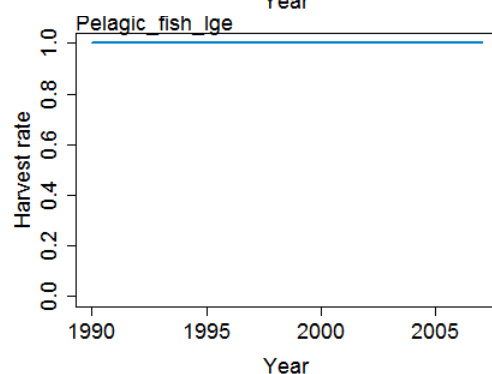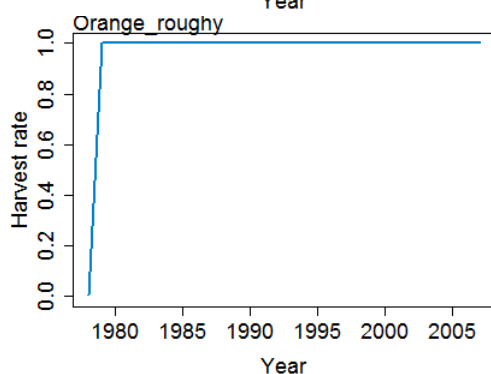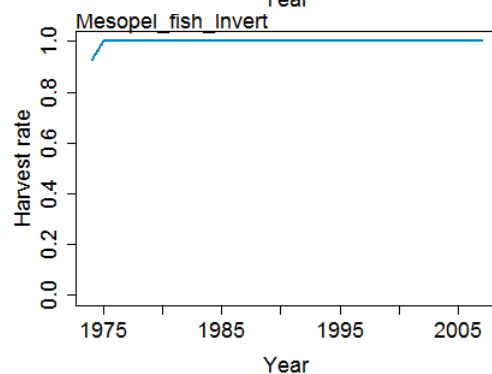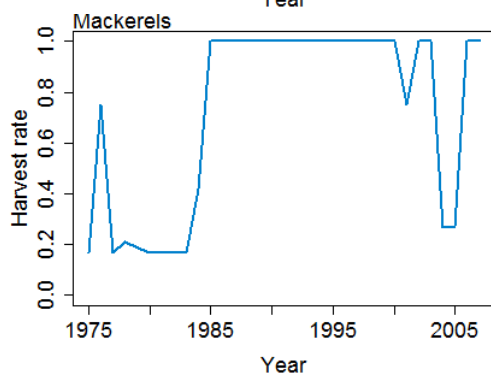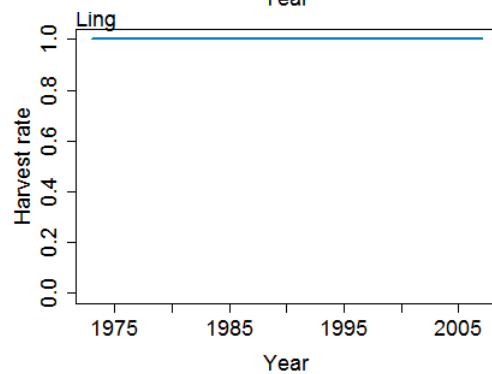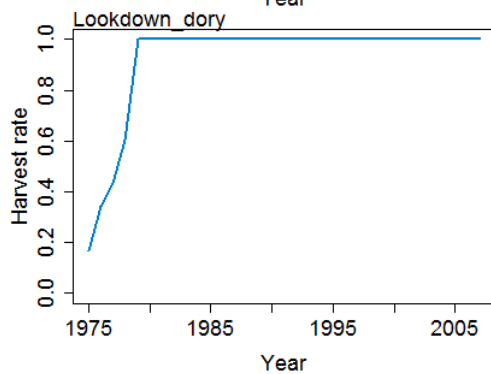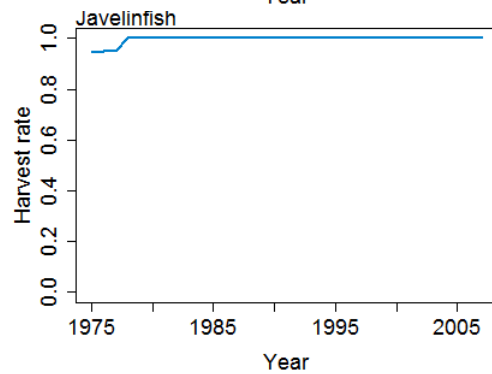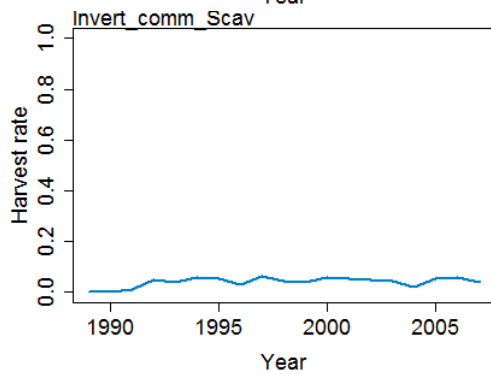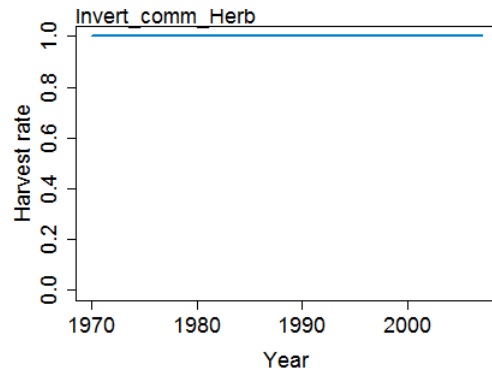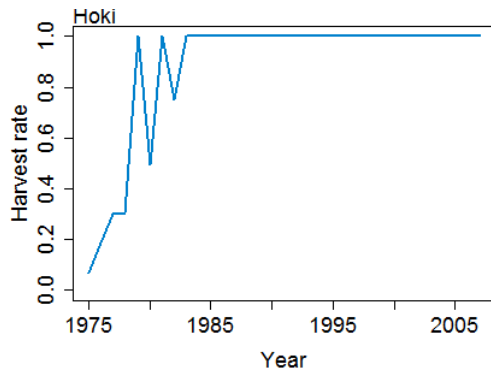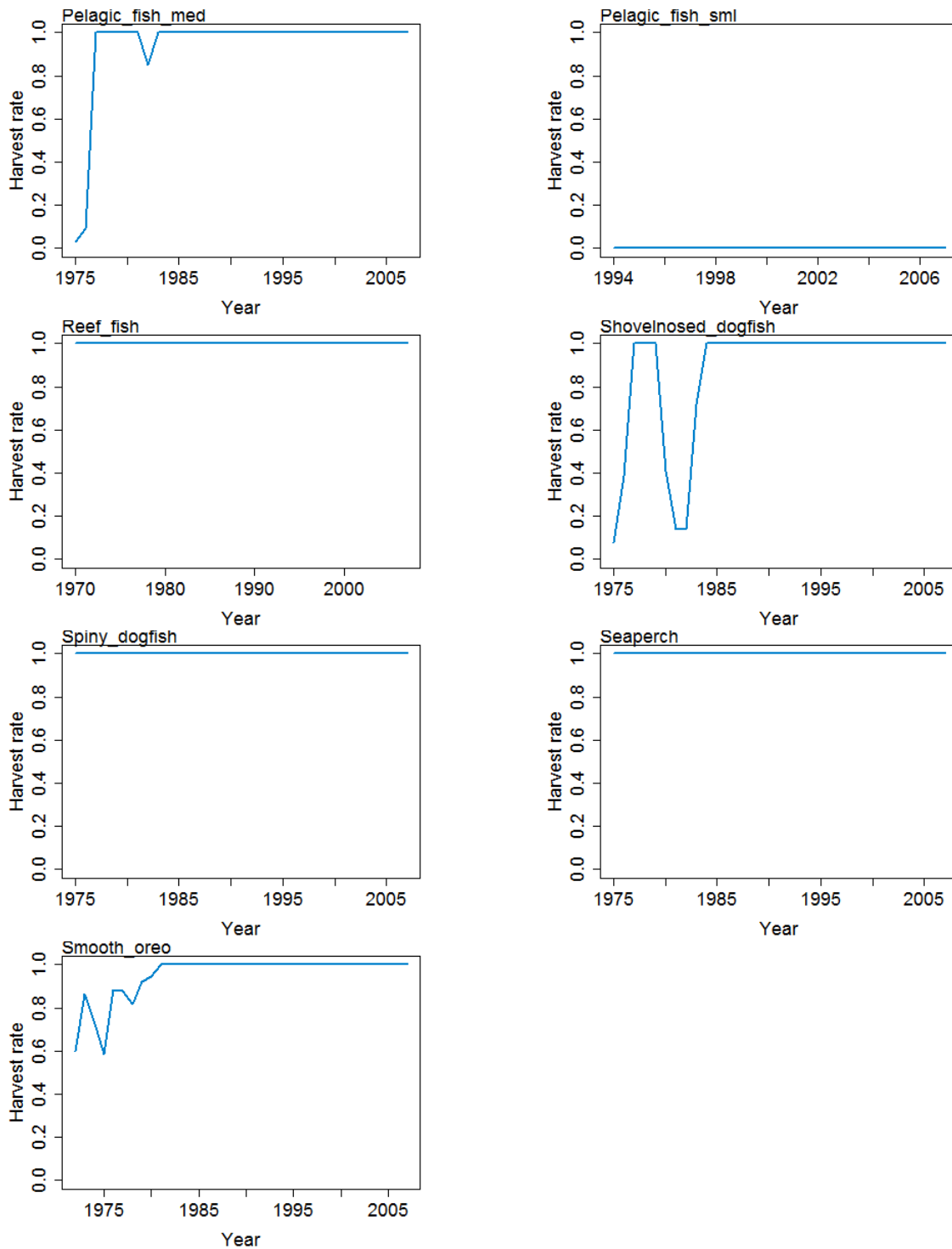
Figure 9.12. Harvest rate from the model output for the 34 species groups that are fished.

The Harvest Rate indicator is shown in Figure 9.12. The dynamics of many of the species groups in the current model are unrealistic and hence these groups have responded to fishing pressure in unrealistic ways. Many of the populations in the model became much lower than they should be, which resulted in HR close to one, and some became over abundant resulting in HR close to zero. Of those that

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responded somewhat realistically, black oreo harvest rate was around 10%, benthic fish deep 40%, basketwork eel 40%, epibenthic fish deep 5%, and invertebrate scavengers 5%.

Shannon's Diversity Index (SDI) sits at around 3 for the active historical fishing period (Figure 9.13) and mean trophic level settled at approximately 1.6.
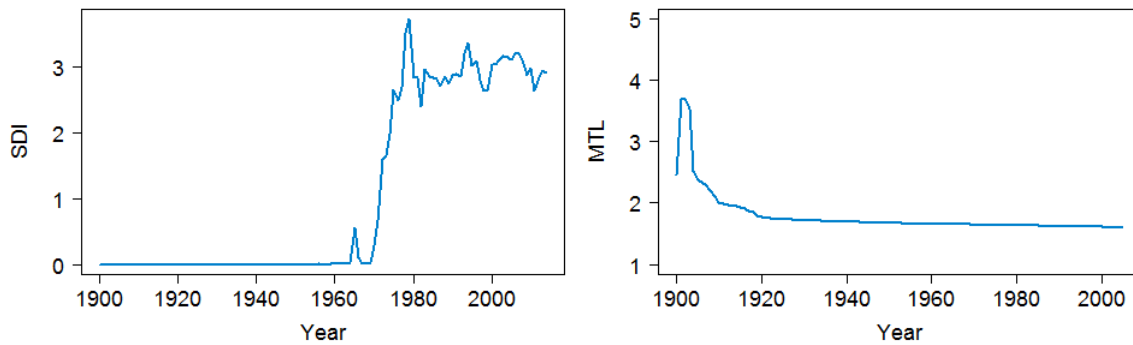
Figure 9.13. Good environmental status indicators: Shannon diversity index (SDI) and mean trophic level (MTL).

Economic indicators

The economic indicators, revenue, cost and profit are shown for the Chatham Rise hoki targetting fishery in Figure 9.14. As discussed above, revenue has only been estimated on the basis of hoki, hake and ling, and so is not complete, but will reflect the main patterns in the economics of the fishery. Revenue peaked in the late 1990s, but declined rapidly, reached a recent minimum in 2006, increased ahgain to 2008, and had declined more gradually in recent years. Costs peaked in the late 1990s, and have fluctuated around a declining trend since this time. Estimated profits declined rapidly after the late 1990s, and were negative up until 2005, but have shown an increasing trend since this time.
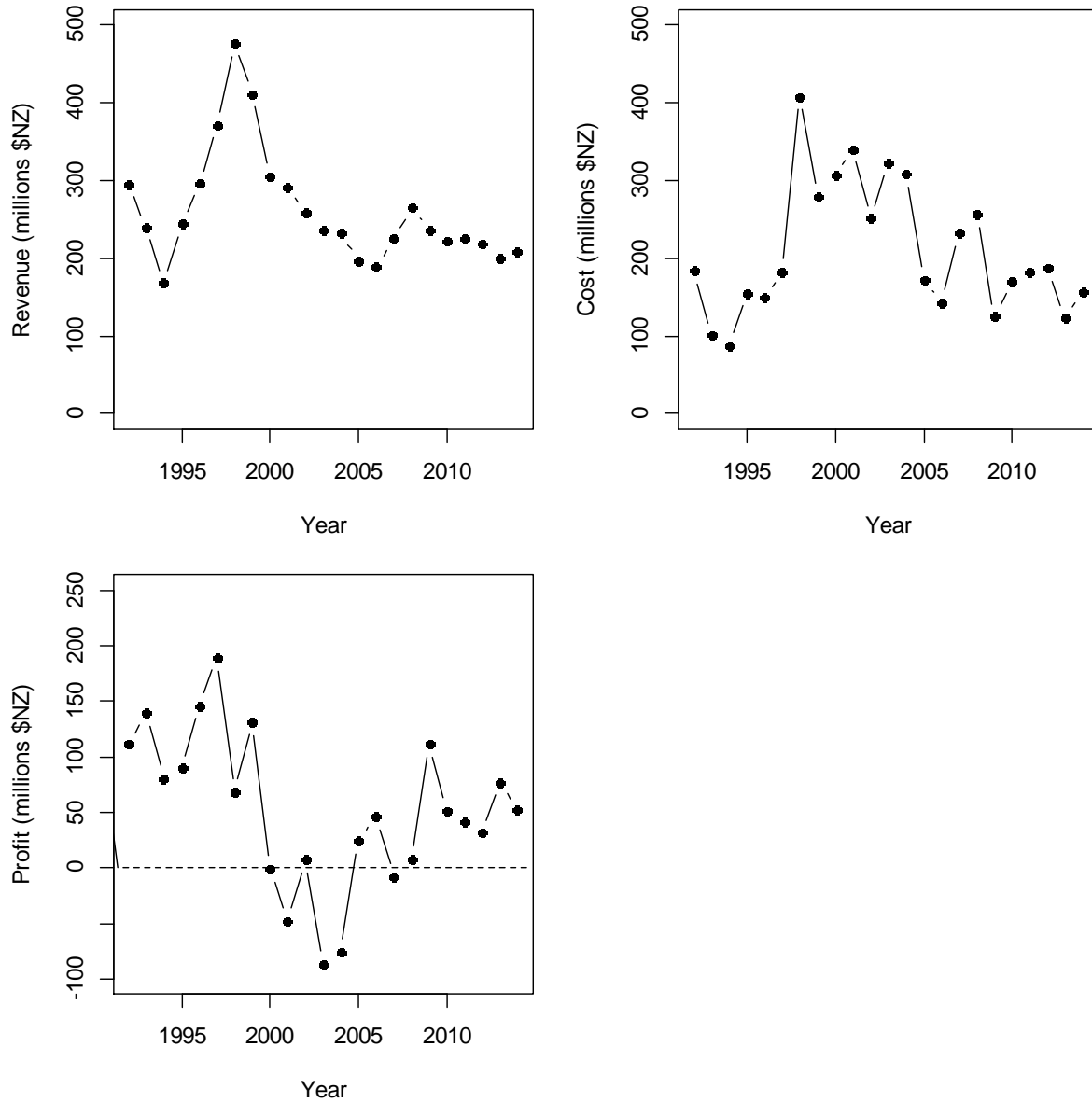
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Figure 9.14. Revenue, costs and profit from the hoki targeting fishery on the Chatham Rise.

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Social indicators

The social indicator used in this case study is the total salaries paid to vessel crew. These salaries are estimated as a 30% of total vessel running costs. The trend (Figure 9.15) largely reflects the changes in fishing effort in the fishery, and shows a steady decline from a peak in the late 1990s.
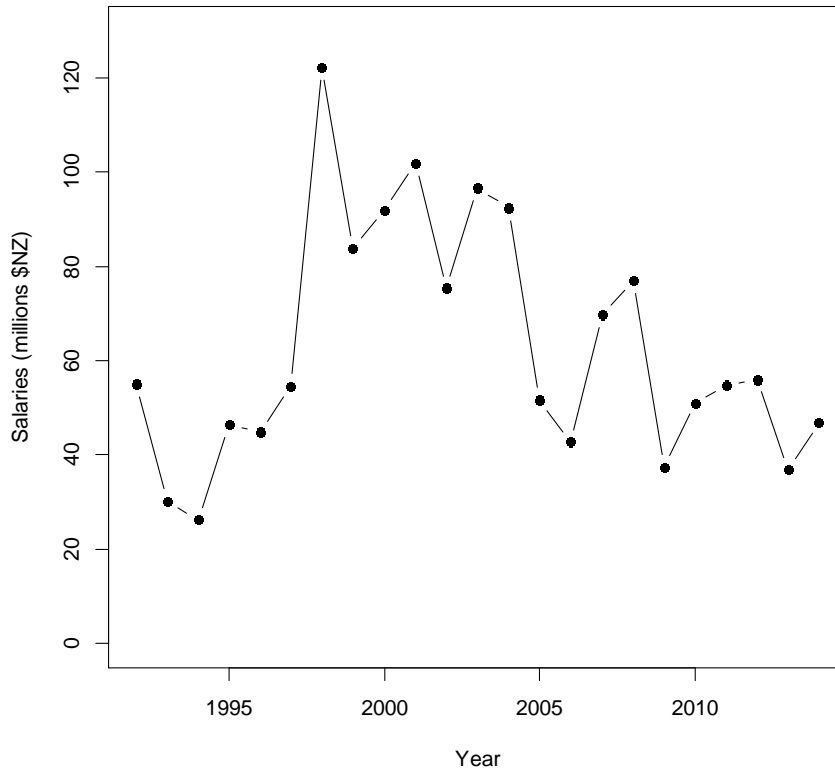
Figure 9.15. Total salaries for the Chatham Rise hoki targeting fishery.

8.4. Conclusion

An Atlantis model has been built for the Chatham Rise region that is able to simulate reasonably realistic biomass and landing trends for some important commercial groups. A range of ecosystem, economic and social indicators have been calculated to help identify how the performance of the main fishery and ecosystem have changed over time. These could also be used to compare management or environmental scenarios. As discussed elsewhere, building a model like the Atlantis model is a continuous process as the model can always be improved. The current model would benefit from further calibration of parameters but it still shows realistic output and can be used for scenario testing. A model like this has a great potential to explore effects of fishing and other drivers on the whole ecosystem and hence support the ecosystem approach to fisheries management.

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This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no. 613571

9. BLACK SEA CASE STUDY: GADGET

9.1. Introduction

Black Sea ecosystem is seriously affected by dynamic changes directly related to fishing, climate change and pollution. Fishery is the most affected sector by the changes of the Black Sea ecosystem. In the same time, fishing activities contribute themselves to the worsening of the ecological situation and for the depletion of the fish stocks. The objective of the Black Sea case study is the restoration of turbot fisheries to more productive levels, considering both the effect of fisheries and the ecosystem change that has occurred in the last 30 years.

The ecosystem models employed in this case study are GADGET and Ecopath with Ecosim (EwE), with the aim of increasing the knowledge about the Black Sea ecosystem functioning and thereby serve to advise on the rebuilding of the turbot stock. These models will allow providing input to the development of a management plan.

9.2. Food web

EwE is designed for construction, parameterization and analysis of mass-balance trophic models. The EwE model developed in this CS considers 10 species or pool of species (turbot, anchovy, sprat, whiting, gobies, mussel, cetaceans, zoobenthos, zooplankton, and phytoplankton). Trophic relationships are modeled with a diet matrix representing the proportion of a prey in the diet of the predator. Other data used in the EwE model are: biomass (t/km2), commercial landings (t/km^2/year), IUU catches (t/km2/year), P/B= Z (total mortality), Q/B (consumption rate).
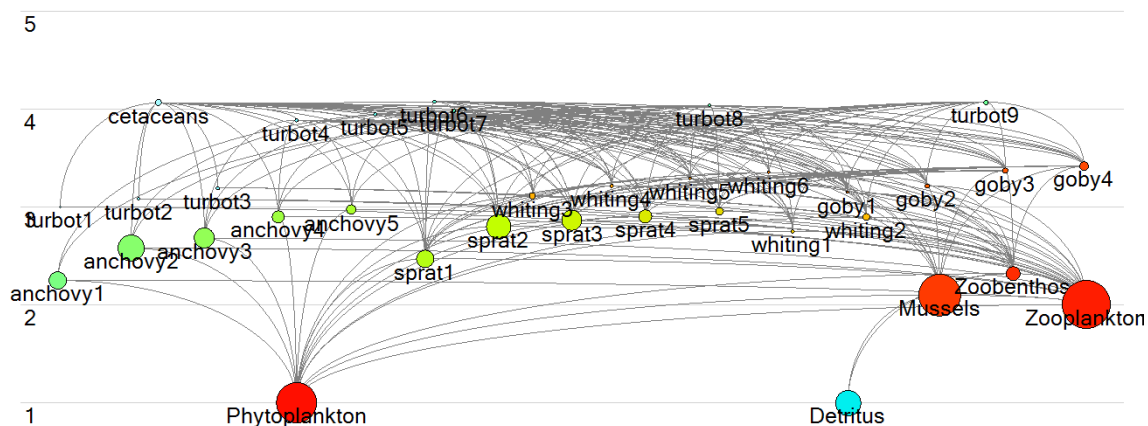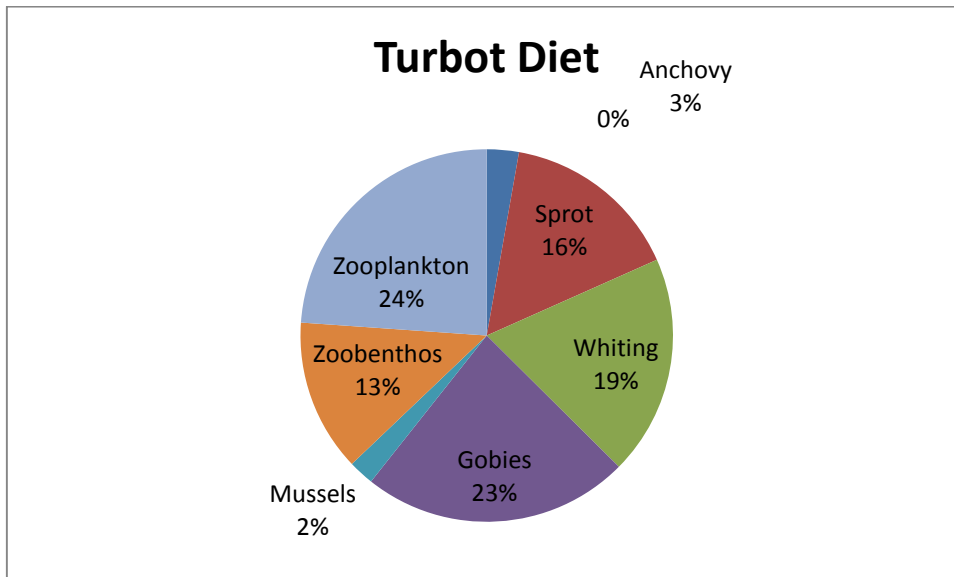

Figure 1. Flow diagram

Diet

Turbot diet novel data obtained from a stomach content analysis has been implemented in a multispecies model in the area for the first time. A protocol for this implementation is presented here.

Figure 2. Turbot diet

Turbot is supposed to feed mainly on gobies, horse mackerel, crustaceans and molluscs (Bănaru et al., 2009). However, the analysis performed shows that gobies was the main prey (Fig. 2). These modifications in diet can reflect changes in the availability of prey which influenced fish diet composition and were probably related to the lost of biodiversity in the Black Sea benthic communities, which became dominated by some opportunistic species (Bănaru et al., 2009). The favourite prey for turbot at age 1 and age 2 is zooplankton, for turbot at age 3 it is zoobenthos, for turbot at age 4 to 7 it is gobies, for turbot at age 8 and age 9 it is sprat, whiting and gobies (Table 2). So, generally, the favourite prey are gobies.

9.3. Methods

Stomach content data are used to define the turbot food web structure in the EwE model in the Black Sea. Ecopath parameterizes models based on two process, one to describe the production and one to describe the energy balance of each group. Ecopath bases the parameterization on an assumption of mass balance over a year creating a static mass-balanced picture of the resources in an ecosystem and their trophic interactions. Once Ecopath has been built, it can be used directly for dynamic modelling using Ecosim.

Modelled species (or pools) are further split into ontogenetic groups as follow: 9 turbot age groups, 5 anchovy age groups, 5 sprat age groups, 6 whiting age groups, 4 gobies age groups, mussel, cetaceans, zoobenthos, zooplankton, and phytoplankton. Trophic relationships are modelled with a diet matrix, i.e. the proportion of a prey in the diet of the predator .

9.4. GES indicators

1) The Shannon's diversity index(SDI)

SDI = $- \sum_s *[P_s * \ln(P_s)]$, where P_s is proportion in mass of species s in the yearly total landings.

2) The large fish indicator (LFI) reflects the size structure of the fish assemblage, which is assumed to be primarily affected by size-selective exploitation but is mediated by species composition as well as the fishing-induced reduction of life expectancy of each exploited species. This indicator was calculated as:

LFI = $W_{>40\,cm}/W_{total}$, where $W_{>40\,cm}$ is the weight of fish >40 cm in length and W_{total} is the total weight of all fish in the survey

3) The mean maximum length of fish (MML) reflects the species composition of a fish assemblage, where fishing is expected to cause a decrease in the proportion of species with large asymptotic body size, slow growth rate, late age and large size at maturation . This indicator was calculated based on the asymptotic total length of each species as:

MML = $\sum(W_s * L_\infty)/\sum W_s$, where W_s is the total weight of species s caught during the survey.

4) The mean trophic level (MTL) of all fish caught during the survey indicates the effect of fishing on the food web. It was calculated as:

MTL = $\sum(TL_s * W_s)/\sum W_s$, where TL_s is the mean trophic level of species s (from Fishbase) and W_s is the total weight of species s caught during the survey.

5) The marine trophic index (MTI) reflects the trophic structure of the fish assemblage where fishing is expected to affect mostly the upper part of the food web, that is, predatory fish. It is defined as the mean trophic level of predatory fish caught during each survey, taking into account only species whose trophic level is higher than or equal to 3.25.

6) Pelagic to demersal fish ratio

P/D = $\sum B_P/\sum B_D$, where B_P = biomass for demersal species, B_D = biomass for pelagic species

7) Harvest rate

HR = C_i/B_i, where C_i = total catch for year i, B_i = total catch for year i

9.5.Results

a) Stock-based indicators

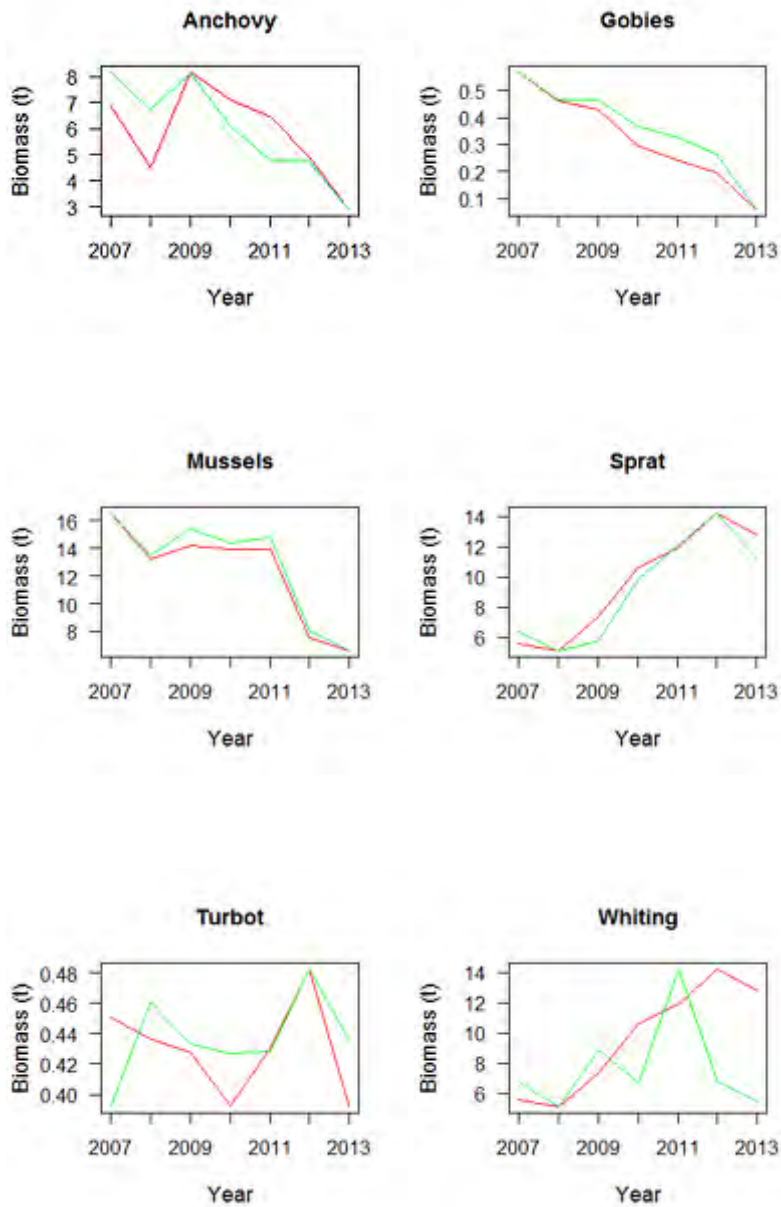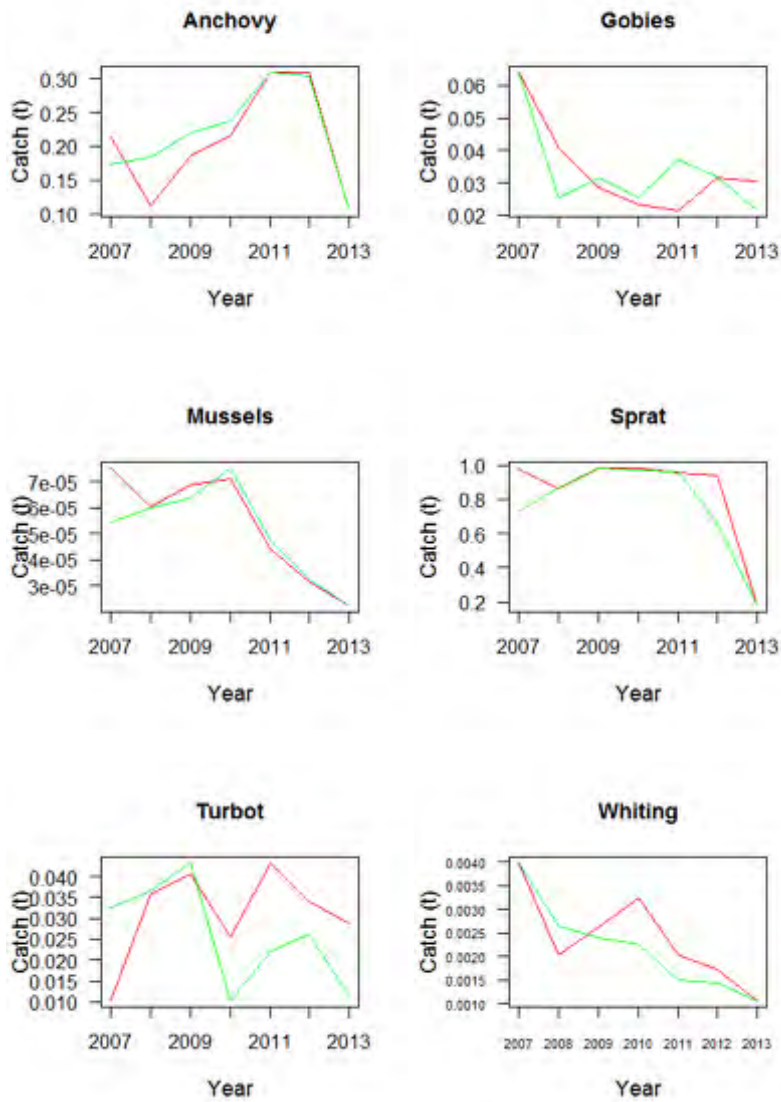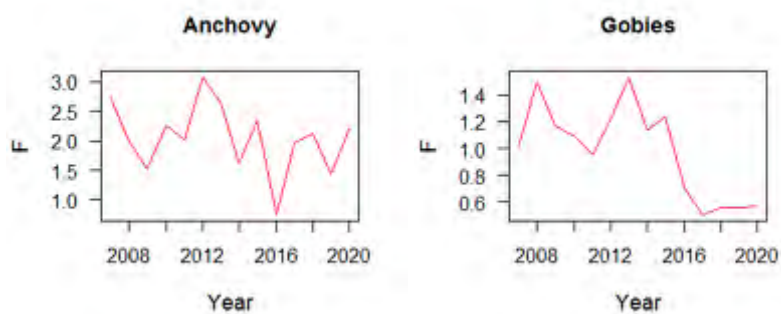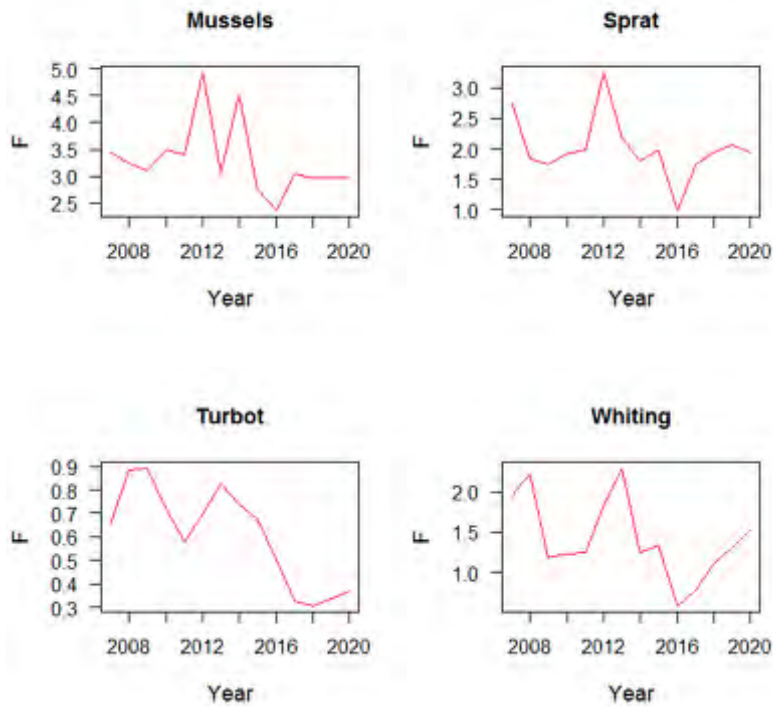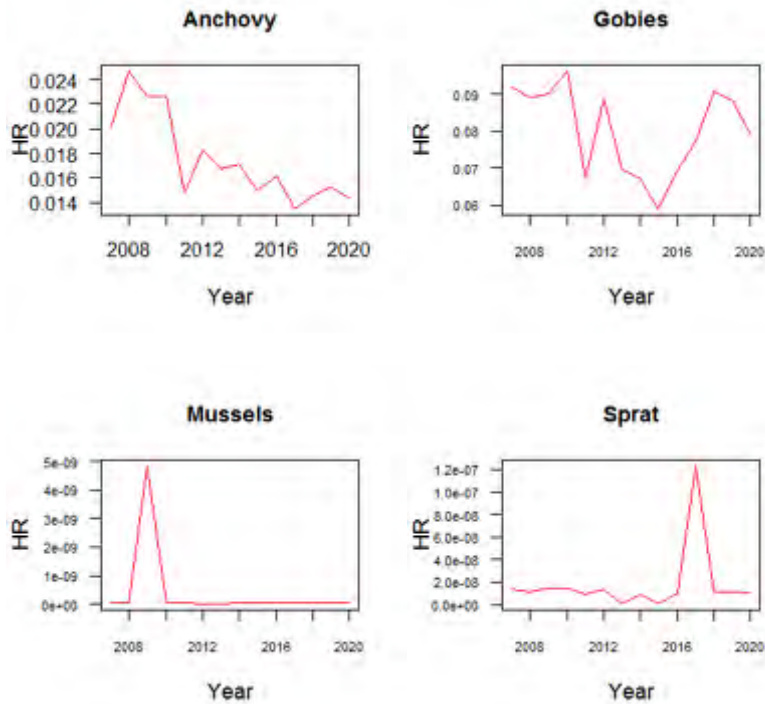**Figure 1**. Biomass modelled(red line) and estimate(green line)
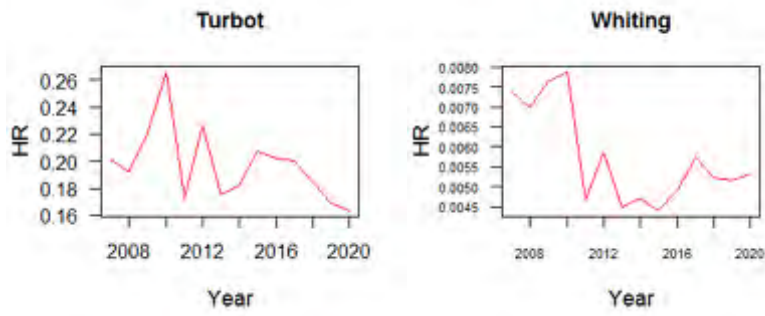
Figure 2 Catches modelled and estimate

Figure 3. Fishing Mortality

b) Good environmental status indicators(GES indicators)

MareFrame

This project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no. 613571
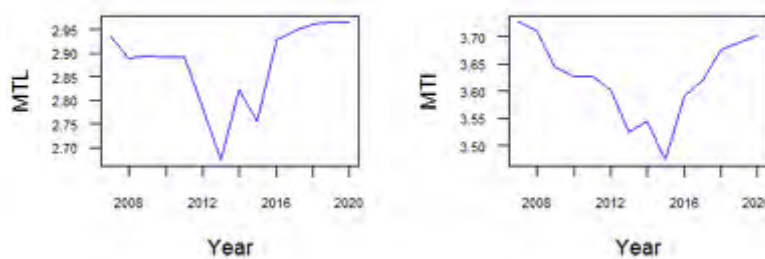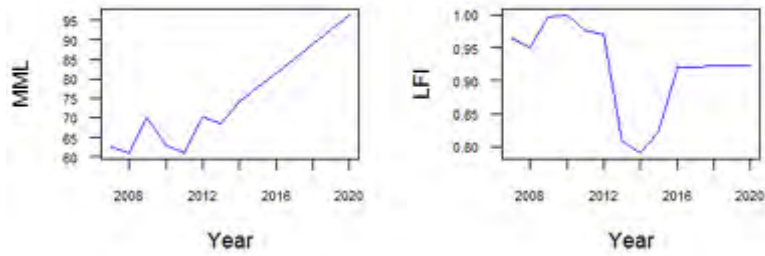
Figure 4. Harvest rate
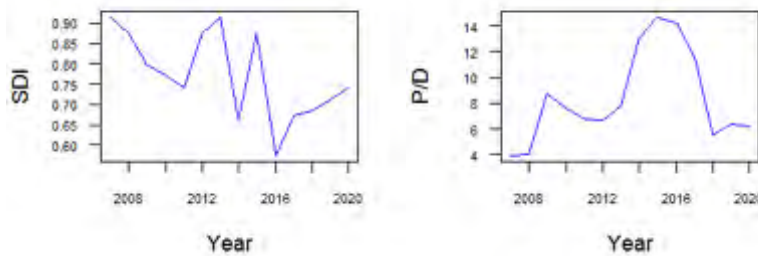
Figure 5. Shannon's Diversity Index and Pelagic to Demersal fish ratio

- Trophic-based indicators

Figure 6. Mean Trophic Level and Mean Trophic Index

- Size-based indicators

Figure 7. Mean Maximum Length and Large Fish Indicator

10. DISCUSSION AND CONCLUSION

All case studies have provided **alternative model** runs. The actual implementations have varied quite a bit, in accordance with the varying emphases in the very different case studies. The alternative models are in most cases quite different in nature from the primary models. For example, a Gadget model has at its base a formal statistical methodology to estimate unknown parameters whereas Atlantis has at its core a whole-of-ecosystem simulator. It has, in all case studies, been a very useful exercise to test more than one model. This "usefulness" is found first as a result of testing out more than one modelling framework, thus forcing the users to look at more than one aspect of modelling the ecosystem. But it also provides multiple bases to compare the various ecosystem models.

Upon completion of this deliverable it is now feasible to undertake several model comparisons which have not been so generally available before. Thus from D4.3 and D4.6, model runs are now available across multiple ecosystems permitting across-ecosystem comparisons and further, the Atlantis framework has now be used to compare other ecosystem models through it use an operating model to simulate data, as was done in D4.4 and will be done to estimate the effects of poor data on ecosystem models in D4.7.

Finally, having the multiple models facilitates comparisons on the likely usefulness of the different models in different environments.