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Deliverable D5.3

Results of ecosystem modelling in case study areas

31/12/2016

Executive Summary

This report is a deliverable of Work Package 5 (WP5 – Apply new methods in case studies) of the FP7 MareFrame research project. It documents the application of a large number of ecosystem models into the different case studies with the ultimate goal to provide the quantitative information required by the decision support tools developed in WP6.

For the purposes of this deliverable, the case studies are developed independently from each other and are reported as separate experiences. However, they share large part of the methodologies (ie., a pool of state-of-the-art ecosystem models) and they all rely on considerable interaction with stakeholders groups. In addition, all the case studies aimed to apply multiple ecosystem models and their implementation has been synchronised aroud case study specific fisheries management issues.

The Baltic Sea case study investigates the management of cod, herring and sprat fisheries in the central Baltic considering trophic interactions among these three stocks, and the uncertainty associated to climate and nutrient scenarios. The case study applied three complementary ecosystem models - ie. EwE, Gadget and a multispecies production model - with the intent to better evaluate the implications of alternative fishery management strategies at population, community and ecosystem level and to serve from tactical to more strategic decisions. The models share common key datasets managed through a case study database. They have been aligned to implement a minimum number of common scenarios, some of which are still under implementation, and their output are expected to be integrated into a decision support tool developed in co-creation with the stakeholders.

The North Sea stakeholders asked for a multispecies model to answer their concerns. This request fitted well with the aim of MareFrame project of removing barriers that currently prevent a more widespread use of an Ecosystem-based Approach to Fisheries Management (EAFM). The so called 'Green model' that was developed to meet this need is a front end model that emulates the results of more complex models using simple approximations and also builds on to these the required social, economic and GES outputs. This results in a model that is extremely transportable and stakeholder friendly, and which is able to highlight the various trade-offs of fisheries management actions. It can take results from a number of pre-existing and developing models for the North Sea. These include the pre-existing and well-reviewed SMS, EwE and Ensemble models together with various developments of the Charmingly Simple Model (CSM), the Multispecies Schaefer model and a proposed delay difference model. More area and fishers behaviour explicit models (the Amber and Red models) are also planned to address further stakeholder concerns. Collectively this wide range of models provide strength through diversity and complementarity.

For Icelandic waters there exist management strategies in the form of a quota system and catch rules for most commercial species, but all of this is in a single-species context. The stakeholders did not show much interest in changing the actual implementation, although quota allocations remain a concern. The most important scientific issues are to consider the effects of multispecies and technical interactions and whether the single species harvest strategies are still sensible in a multispecies context.

The west of Scotland faces several management issues: the stocks of cod and whiting in ICES area 6a are currently depleted and the population of grey seals, and consequently the predation mortality on gadoids, has been increasing for the past two decades. In collaboration with the stakeholders group a set of alternative management strategies has been identified to address these issues. These alternatives were modelled with the Ecopath with Ecosim ecosystem model for the west of Scotland which was updated for the occasion using the latest assessment and survey outputs. In addition, Good Environmental Status and socio economic indicators were computed from the model outputs to assess the performance of the alternatives strategies regarding ecosystem health and fisheries economy. Results showed the importance of considering trophic interactions when assessing different fishing scenarios. Applying single species FMSY values seems insufficient to bring cod within safe limits by the end of the 20 years simulation period. Unsurprisingly, the alternative with the lowest fishing mortalities across species returned the highest ecosystem indicators overall but resulted in the lowest biodiversity. Increasing fishing mortality on crustaceans and pelagic species increased profit in the short term but not on the medium and long term. All the alternatives tested, including the ones with the lowest and highest fishing mortalities, converged towards similar long term total profit at the end of the simulation period.

Two different model approaches were developed in the South Western Waters. The first consists in a bioeconomic model for the anchovy fishery in the Gulf of Cádiz. The main objective for this is to provide advice for an adaptive management (considering environment forcing) in contrast to the current management based on a fixed quota. The main environmental processes considered were sea surface temperature, Guadalquivir River discharges and intense easterly winds. Scenarios with fixed and adaptive TACs were tested. Adaptive TACs provides a higher stability in catches and profits with a reduced probability of collapse. The second consists a multispecies model for hake and dolphins in the Iberian Peninsula. The main goal is to explore management measures considering hake and dolphin targets. Stakeholders suggested scenarios to explore the consequences of delaying the achievement of MSY until 2020. This delay produces a higher stability in catches and revenue although reduces future yield and the recovery for hake and dolphins.

The Strait of Sicily case study focuses on the development of a tool to support the implementation of an ecosystem approach to fisheries management (EAFM) in a key fishing area of the Mediterranean Sea. The objectives of the case study have been progressively refined through the application of a co-creation approach with key stakeholders (i.e. fishers and fishers representatives, managers of local and national administrations, conservation NGOs, FAO and GFCM officers) and taking into account the objectives of the GFCM international management plan for bottom trawl fisheries exploiting deep water rose shrimp (DPS: *Parapenaeus longirostris*) and hake (HKE: *Merluccius merluccius*) in GSAs 12-16. The overall goal of the case study has been adapted to provide a Decision Support Tool for the application of EAFM in the area which could support the achievement of long term sustainability finding a balance between ecological and human well-being through good governance. Atlantis and Gadget ecosystem models have been developed to provide advice on the effects of different management scenarios in relation to the following four main management objectives identified in co-creation with the stakeholders: i) rebuilding overexploited stocks; ii) long-term continuity of the fishing activities; iii) same rules for all; iv) good environmental status.

Ecosystem modelling in the Black Sea case study was made using Gadget and EwE ecosystem models. The Gadget model includes 7 different populations or functional groups, with 3 fleets acting in 3 different areas (Romanian area, West Black Sea area and all Black Sea). Both the impact of the interactions between species and the impact of fisheries harvesting the species have been included in the model, and the model successfully reproduces trends in historical data. The EwE model considers 10 species or pool of species. Trophic relationships are modeled with a diet matrix representing the proportion of a prey in the diet of the predator. Both models indicate a sharp decline of the total turbot biomass, even when eliminating IUU, suggesting that a temporary ban on turbot fisheries is among the management measures to be considered.

A balanced foodweb model has been developed for the Chatham Rise case study area, and used to investigate the potential implications of seabed mining for phosphorite nodules. The Chatham Rise is a very important area for New Zealand commercial fisheries, and also supports a diverse seabird communities, marine mammals and deepwater corals. Proposals have recently been developed to mine the seabed within an area currently protected from benthic disturbance. The foodweb model was used to the trophic importance of different components of the foodweb, and then on the basis of expert knowledge of the life history of the different components, those most likely to be sensitive to the effects of mining, and the potential implications were identified. These results were used to inform the New Zealand Environmental Protection Agency in consenting decisions on the seabed mining application.

Contents

Introduction

Models developed in WP4 are implemented in each case study area to investigate the effect of fishing and climate change scenarios on key ecosystem processes. A selection of those models provides a basis for the development of decision support tools in WP6.

This deliverable summarises in the following eight chapters the application of ecosystem models in the eight MareFrame case studies: (1) the Baltic Sea, (2) the North Sea, (3) Iceland waters, (4) West of Scotland, (5) Iberian waters, (6) Strait of Sicily, (7) Black Sea, and (8) Chatham Rise. For the purposes of this deliverable, the case studies are developed independently from each other and are reported as separate experiences. However, they share large part of the methodologies (ie., a pool of state-of-the-art ecosystem models) and they all rely on considerable interaction with stakeholders groups.

1. Baltic Sea case study

Brief description of the case study objectives

The case study approaches the cod, herring, and sprat fisheries in the ecosystem context in the Central Baltic. There exists rather good biological knowledge about these species and the trophic interactions in the 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.

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

Modelling framework

We used three main modelling frameworks to the Baltic Sea case study: Ecopath with Ecosym (EwE), Gadget and a multispecies production model (MSPM). The three frameworks have been selected for a number of reasons. EwE has been proved in the past to be able to reconstruct the flow dynamics across the main functional groups and species in the Baltic Proper and the regime shift which occurred in the late 1980s (Tomczak et al. 2012). MSPM has been used in central Baltic to capture trophic interactions between cod, herring and sprat (Horbowy 1995, 1996). Gadget has been used succesfully in other systems to model trophic interactions among a small number of species (Perez-Rodriquez et al. 2016) and where information on age is uncertain but length data are available (Taylor et al. 2007) and for this reason is expected to be suitable for purposes of the Baltic Sea case study. The three models offer an interesting level of complementarity in the analysis of ecosystems which has never been evaluated, and ultimately expertise on these three frameworks existed within the case study. This deliverable (D5.3) presents the model development which has been independent within each modeling framework but coordinated in two main aspects: the use of a common dataset (see D2.2, D2.4), organised with mfdb into a case study database (see WP3), and in relation to the management objectives and scenarios for the case study both developed in co-creation with our stakeholder group.

Ecopath with Ecosim (EwE)

Conceptual model

The Ecopath with Ecosim food-web model of the open Baltic Sea used in the case study is the further development of an earlier model (Tomczak *et al.*, 2012; Niiranen *et al.*, 2013). It represent biomass flows among organismal groups within the food web and to fisheries in the 'model year', 2004. In contrast to the model of Tomczak *et al.* (2012), the model used in the case study was parameterised according to post-regime shift conditions. The functional groups in the model represent the most important groups in the offshore central Baltic Sea. The model includes charismatic species such as grey seals and offshore fish-feeding birds, the four commercially most important fish species, the benthic part of the food web and 4 zooplankton groups (Figure 1).

Primary producers are represented by one functional group – phytoplankton, which we considered the most appropriate as the model is working in annual time steps. Thus, the phytoplankton group in the model reflects total standing stock of pelagic primary producers and their production. Mesozooplankton was divided into four functional taxa-related groups: *Pseudocalanus* spp., *Acartia* spp., *Temora* spp., and other mesozooplankton, which consists of other copepods and cladocerans. The first three species-related functional groups were chosen to represent key species in the pelagic part of the food-web, with important role in shaping the energy transfer due to their sensitivity to climate change and trophic cascades as well as by influencing fish recruitment processes (Möllmann *et al.*, 2008; Casini *et al.*, 2009). Mysids, which are an important food item of fishes, were included as a single group (Casini & Cardinale, 2004). The benthic community was split into five groups, *Saduria entomon, Macoma balthica, Mytilus* spp., meiobenthos and 'other macrozoobenthos'. The explicit representation of some species related functional groups within the macrozoobenthos was done to reflect their essential roles in the diet of benthic feeding fish (cod and flounder) and significant share in total macrozoobenthos biomass. Meiobenthos functional group represent processes of recycling within sediment (Witek, 1995; Harvey *et al.*, 2003) while other macrozoobenthos are biomass of other species in the benthos community.

There are four functional groups of fish – sprat (stock at ICES SD 22-32), herring (Central Baltic Herring stock ICES SD 25-29;32 ex GOR), cod (Eastern Baltic cod stock ICES SD 25-32) and flounder (stock ICES SD 24-25, 26-28 and 27-29) as these are the biomass dominating and commercially important fish species in the Baltic Proper.

Grey seals and fish eating birds represent top predators. Seals are the top trophic level of Baltic food-web and play a role of top-down factor for fish population. Due to significant increase of seal abundance in the Baltic (Härkönen *et al.*, 2013) and rising conflict with fisheries it was important to include this group in the model. Fish eating birds were usually neglected in the open Baltic food-web models (Harvey *et al.*, 2003; Tomczak *et al.*, 2012). We decide to include them in order to better reflect their effects on fish as well as to represent an important link to the terrestrial ecosystem. Fish eating birds ate also an indicator group of ecosystem health related to fish condition (Österblom *et al.*, 2006, 2007). Due to the landings obligation and discard ban in the Baltic Sea some species of fish eating birds might be affected by reductions of easily available food discarded from fishing vessels. That hypothesis was also one of the reasons to include the group in the model.

We considered to include also alternative functional groups such as i.e salmonid fish (*Salmo salar, Salmo trutta*) to represent the economically important group of fish species. However, because of their overall low biomass at open sea and seasonal migratory behaviour we decided not to include them. Harbour porpoise (*Phocena phocoeana*) is one of the marine mammals inhabiting the Baltic Sea, however due to its very low biomass density and marginal consumption impact on fish stocks, we also decided to omit this species. Other fish species such as perch, pike, pike-perch and sticklebacks and birds like cormorants are mainly associated with Baltic costal zone ecosystems ware not included.

All modelled fish groups were represented as 2 life stages, adult and juvenile. Multistanza representation of life stages enables the model to reflect ontogenetic changes in diet. The effects of fishing on the food web are represented by fishing mortality, which in Ecopath is calculated as (landings+discards)/biomass. Landings and discards were set based on data from the ICES Baltic Fisheries Assessment Working Group , except discards of flounder where we used data reported by the European Commission's Scientific, Technical and Economic Committee (STECF).

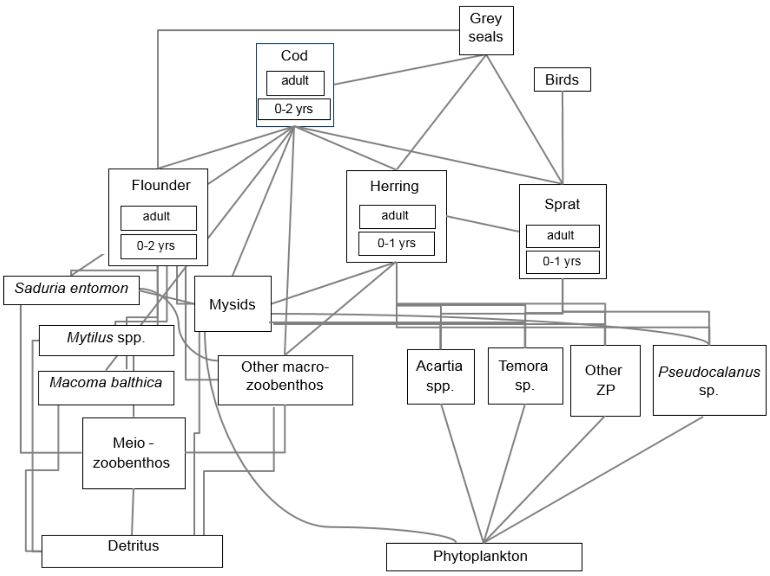


Figure 1.1 Functional groups and their feeding relationships in the EwE model.

Model selection and description of best model- Ecopath

EwE models have two main components: the static Ecopath model describing yearly biomass flows which serves as initial condition for the dynamic Ecosim. For the Ecopath component the most important criterium for model selection is that parameters are chosen in such a way that biomass flows in and out of each group are balanced. Besides this condition, there are a set of quality criteria and established 'best practice' recommendations for EwE models that were followed when parametrising the model (Heymans *et al.*, 2016).

Details of the final parametrisation of the Ecopath model are described in Table 1.1, Table 1.2 and Table 1.3 Diet proportion of prey in weight for each predator-prey pair in the model.

Table 1.1: Ecopath input parameters, references and assumptions or changes implemented compared to the references, when applicable. All biomasses (B) are in units of t/km^2. 'Total mortality' parameter of multistanza groups is equivalent to P/B (production/biomass) of other groups. 'Q/B' refers to consumption/biomass, 'UA' unassimilated consumption', 'BA' to biomass accumulation rate and 'DC' to diet composition.

Group name	parameter	value	source	comment
Grey seal	*B*	0.006	BALSAM Grey Seal Database (cross-checked with: Lundström et al. *in press*)	BALSAM reports estimates on number of seals in various areas. Numbers in areas contained in the Baltic Proper were summed. Numbers were converted into density by assuming an average seal weight of 100 kgs and an area of 240 000 km^2.
	P/B	0.1	Harvey *et al.*, 2003	
	Q/B	16.28	Gårdmark *et al.*, 2012 (cross-checked with: Lundström et al. *in press*)	Both sources report daily food consumption in kg food, calculated to yearly amount and assuming a 100 kgs of average seal weight.
	DC		Lundström et al. *in press*	
Fish-feeding birds	*B*	0.002	Durinck *et al.*, 1994; Österblom *et al.*, 2002	Razorbill and black guillemot abundances from the early 90's were reported in Durinck *et al.*, 1994; were converted to densities assuming weights of 700 and 400 g-s, resp, accounting for an estimated increasing trend in the populations to the 2000's reported by Herrmann et al., (2013). Common guillemot abundance estimates are by Österblom et al., (2002) are

				similarly converted to densities, assuming an average weight of 1 kg.
	P/B	0.1	Harris *et al.*, 2000; Lavers *et al.*, 2009	Calculated as equal to mortality (1-survival rate).
	Q/B	130	Lilliendahl and Solmundsson, 1997 Mehlum and Gabrielsen, 1993	Estimated based on daily food intake and body mass values reported for razorbill (the most abundant species in the group). Corresponding estimates for common guillemot are 170 (Enstipp *et al.*, 2006) and for black guillemot 223 (Mehlum and Gabrielsen, 1993), the latter likely being an overestimation as it is based on food intake values during the chick-rearing period.
Adult cod	*B*	0.33	ICES, 2013	Value between SSB and Age3+ biomass from SAM model output for Eastern Baltic cod (SDs 25-29, excl. Gulf of Riga). Area used for calculating density is 240 000 km^2.
	Total mortality	0.885	ICES, 2013; FishBase	Total mortality calculated as the sum of natural mortality M (0.18, FishBase, value from Gdansk Deep) and fishing mortality in 2004 calculated as: (landings+discards)/B.
	Q/B	3.81	Witek, 1995	
	UA	0.17	Harvey *et al.*, 2003	
	DC		Huwer *et al.*, 2014; ICES, 2016	Average diet 2003-2005 of cod>=33 cm, stomachs collected in SD 25-29. Also see note below.
Juvenile cod	*B*	0.354	calculated by EwE	
	Total mortality	1.062		Total mortality assumed to be 1.2 times adult total mortality.

	Q/B	7.65	calculated by EwE	
	DC		Huwer *et al.*, 2014; ICES, 2016	Average diet 2003-2005 of cod<33 cm, stomachs collected in SD 25-26. Also see note below.
Adult herring	*B*	2.33	ICES, 2015	XSA assessment of Central Baltic herring, SDs 25-29 and 32, excl. Gulf of Riga) , Age 2+. Area used for calculating density is 280 000 km^2.
	Total mortality	0.78	ICES, 2015; FishBase	Calculated as the sum of natural mortality M (0.65, FishBase, estimated using life-history tool) and fishing mortality in 2004 calculated as landings/B.
	Q/B	3	Witek, 1995	Witek, 1995 reported 1.96, adjusted to have more realistic production/consumption values.
	DC		Casini & Cardinale, 2004; Möllmann *et al.*, 2004; Tomczak *et al.*, 2012	DC values chosen within the range of DCs reported to satisfy the mass-balance assumption.
Juvenile herring	*B*	1.003	calculated by EwE	
	Total mortality	1.176		Assumed to be 1.5 times adult total mortality.
	Q/B	5.811	calculated by EwE	
	DC		Casini & Cardinale, 2004; Möllmann *et al.*, 2004; Tomczak *et al.*, 2012	DC values chosen within the range of DCs reported to satisfy the mass-balance assumption.
Adult sprat	*B*	2.39	ICES, 2015	XSA assessment of sprat in Subdivisions 22-32, Age 2+. Area used for calculating density is 500 000 km2.
	Total mortality	1.24	ICES, 2015; FishBase	Calculated as the sum of natural mortality M (0.65, FishBase, estimated using life-history tool) and fishing mortality calculated as 2004 as landings/B.
	Q/B	4.63	Witek, 1995	

	DC		Casini & Cardinale, 2004; Möllmann *et al.*, 2004; Tomczak *et al.*, 2012	DC values chosen within the range of DCs reported to satisfy the mass-balance assumption.
Juvenile sprat	*Total mortality*	1.865		Assumed to be 1.5 times adult P/B.
	Q/B	8.9	calculated by EwE	
	DC		Casini & Cardinale, 2004; Möllmann *et al.*, 2004; Tomczak *et al.*, 2012	DC values chosen within the range of DCs reported to satisfy the mass-balance assumption.
Adult flounder	*B*	0.463	ICES, 2012	Estimated based on assessments for SD24-25 and 26, assuming higher density in SD28 (based on BITS survey).
	Total mortality	0.79	ICES, 2016b; FishBase	Calculated as the sum of natural mortality M (0.2, FishBase, value for Baltic Sea SD 22-32) and fishing mortality in 2004 calculated as: (landings+discards)/B
	Q/B	4.21	Witek, 1995	
	DC		Borg *et al.*, 2014	
Juvenile flounder	*B*	0.422	calculated by EwE	
	Total mortality	1.184		Assumed to be 1.5 times adult P/B.
	DC		Aarnio *et al.*, 1996; Nissling *et al.*, 2007; Ustups *et al.*, 2007; Florin & Lavados, 2010	
Saduria entomon	*B*	2	NMFRI- outer gdansk basin, mean 2002-2003 samples, Haahtela, 1990	
	P/B	1.3	Witek 1995	
	Q/B	5	Witek 1995	Changed from 6.51 during Prebal procedure
	DC		Englund et al. 2008	
Mytilus spp.	*B*	10	NMFRI- outer gdansk basin, mean 2002-	

			2003 samples, Darr et al. 2014	
	P/B	1.75	Witek 1995	
	Q/B	8.73	Witek 1995	
	DC		Mackinson&Daskalov 2007	Based on diet of suspension feeders in the North Sea Ecopath model.
Macoma balthica	*B*	45	NMFRI- outer gdansk basin, mean 2002-2003 samples., Darr et al. 2014, Timmerman et al. 2012	
	P/B	0.4	Witek 1995	
	Q/B	2	Witek 1995	
	DC		Timmermann et al., 2012	A mixture of suspension feeding (see DC Mytilus spp) and deposit (detritus) feeding.
Oth. macrozoobentos	*B*	11.385	NMFRI- outer gdansk basin, mean 2002-2003 samples	
	P/B	2	Witek 1995	Assuming that most abundant groups are *Pontoporeia f.* and Polychaetes (e.g. *Harmothoe sarsi*).
	Q/B	10	Witek 1995	Assuming that most abundant groups are *Pontoporeia f.* and Polychaetes (e.g. *Harmothoe sarsi*).
	DC			A mixture of suspension feeding (bivalves, see DC Mytilus spp.), deposit feeding (amphipods), deposit feeding and predation (polychaetes).
Meiobenthos	*B*	6.8	Olafsson&Elmgren 1997	Summer value, conversion factor from shell-free dry weight to wet weight 1:4.
	P/B	6.17	Harvey et al. 2003	

	Q/B	31.17	Harvey et al. 2003	
	DC		Olafsson et al. 1999	
Mysids	*B*	2.16	estimated by EwE	Assuming an ecotrophic efficiency of 0.75 (Niiranen et al. 2013).
	P/B	5	Mohammadian et al. 1997 in Tomczak et al. 2012	Set to lower value than in source (7.3) to have production/respiration ratio<1 and reflect species shift (Ogonowski et al. 2013) from Mysis spp. to Neomysis with lower P/B (Witek 1995).
	Q/B	15	Harvey et al. 2003	
	DC		Tomczak et al. 2012	
Other zooplankton	*B*	4	average NMFRI 2003-2005	
	P/B	20	Niiranen *et al.*, 2013	
	Q/B	100	Tomczak et al. 2012	
	DC			Phytoplankton feeding
Pseudocalanus spp.	*B*	1.93	Average of: 1. BIOR 2. average NMFRI 2003-2005	BIOR data converted to density assuming an average depth of 62 m, average 2003-2005, average spring-summer across Gotland Sea and Bornholm Basin.
	P/B	7	Niiranen et al., 2013; Witek, 1995	
	Q/B	27	Witek 1995	
	DC			Phytoplankton feeding
Acartia spp.	*B*	3.027	Average of: 1. BIOR 2. average NMFRI 2003-2005	BIOR data converted to density assuming an average depth of 62 m, average 2003-2005, average spring-summer across Gotland Sea and Bornholm Basin.
	P/B	20	Niiranen *et al.*, 2013	
	Q/B	83	Witek 1995	

	DC			Phytoplankton feeding.
Temora spp.	*B*	2.271	Average of: 1. BIOR 2. average NMFRI 2003-2005	BIOR data converted to density assuming an average depth of 62 m, average 2003-2005, average spring-summer across Gotland Sea and Bornholm Basin.
	P/B	20	Witek 1995	
	Q/B	83	Witek 1995	
	DC			Phytoplankton feeding.
phytoplankton	*B*	7.05	NMFRI, 2004.	Average value of open water stations (SD 25-26).
	P/B	200	BALTSEM model output, Johannson et al. 2004, Tomczak et al. 2012;	P/B from BALTSEM (Baltic Sea Long-term Eutrophication Model, Baltic Nest Institute) output was calculated as total annual production Gotland Sea (GS) and Bornholm Basin (BN) to total phytoplankton biomass standing stock GS and BN.
Detritus	*B*	1645	E. Gustafsson pers.comm	Based on BALTSEM model output, the sum of detrital POC, DOC (phytoplankton exudates, zooplankton excretion etc.) and benthic (sediment) OC (the largest pool). Assuming a conversion factor from C to wet weight 11.62 (Tomczak et al. 2012).

Table 1.2: Multi-stanza parameters based on reported values for the Baltic Sea in FishBase and literature values.

Multi-stanza name	Cod	Herring	Sprat	Flounder
VBGF K	0.23	0.43	0.51	0.2
Recruit power	1	1	1	1
BA/B	0	0	0	0
Adult stanza start month	36	24	24	36
Wmat/Winf	0.13	0.38	0.26	0.1

Table 1.3: Diet proportion of prey in weight for each predator-prey pair in the model.

Prey \ predator	Grey seal	Birds	JuvCod	AdCod	JuvHer	AdHer	JuvSpr	AdSpr	JuvFlo	AdFlo	Saduria	Mytilus	Mac. b.	Oth. macrob	Meiob.	Mysids	Oth. zoopl.	Pseudo.	Acartia	Temora
Grey seal	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fish-feeding birds	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juv. cod	0.078	0	0	0.015	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ad. cod	0.032	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juv. her.	0.003	0	0.07	0.046	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ad. her.	0.376	0	0.06	0.42	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juv. spr.	0	0.02	0.3	0.12	0	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ad. spr.	0.207	0.93	0.126	0.278	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juv. flo.	0	0	0	0.030006	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ad. flo.	0.031	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Saduria entomon	0	0	0.11893	0.051	0	0	0	0	0	0.083	0	0	0	0	0	0	0	0	0	0
***Mytilus* sp.**	0	0	0	0	0	0	0	0	0	0.453	0	0	0	0	0	0	0	0	0	0
Macoma b.	0	0	0	0.000213	0	0	0	0	0	0.339	0	0	0	0	0	0	0	0	0	0
Oth.macrob.	0	0	0.00937	0.001267	0	0.1	0	0	0.6	0.125	0.59761	0	0	0.1	0	0	0	0	0	0
Meiobenth.	0	0	0	0	0	0	0	0	0.1	0	0.00398	0	0	0.3	0.02	0	0	0	0	0
Mysids	0	0	0.17064	0.039738	0.1	0.25	0	0	0.3	0	0.39841	0	0	0	0	0	0	0	0	0
Other zoopl.	0	0	0	0	0.2	0.15	0.39	0.33	0	0	0	0	0	0	0	0.05	0	0	0	0
***Pseudoc.* sp**	0	0	0.01	0	0.1	0.1	0.15	0.17	0	0	0	0	0	0	0	0.05	0	0	0	0
***Acartia* sp.**	0	0	0	0	0.1	0.02	0.21	0.09	0	0	0	0	0	0	0	0.1	0	0	0	0
***Temora* sp.**	0	0	0	0	0.5	0.37	0.25	0.41	0	0	0	0	0	0	0	0.2	0	0	0	0
Phytopl.	0	0	0	0	0	0	0	0	0	0	0	0.8	0.4	0	0	0.3	1	1	1	1
Detritus	0	0	0	0	0	0	0	0	0	0	0	0.2	0.6	0.6	0.98	0.2	0	0	0	0
Import	0.274	0.05	0.134	0	0	0	0	0	0	0	0	0	0	0	0	0.1	0	0	0	0
Sum	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Table 1.4: Landings and discards of each stanza (t/km^2).

Stanza name	Landings	Discards
Juvenile cod	0.0143	0.0031
Adult cod	0.2750	0.0024
Juvenile herring	0.0202	0
Adult herring	0.3119	0
Juvenile sprat	0.2851	0
Adult sprat	0.9629	0
Juvenile flounder	0.0013	0.0070
Adult flounder	0.1810	0.0920

Model selection and description of best model- Ecosim

The Ecosim model has the balanced Ecopath as initial state and models how the ecosystem changes compared to that state as an effect of forcing. There were two types of forcing applied in our model: fishing mortalities, abiotic forcing and biomass forcing for grey seals (Table 1.5). We applied the same type of abiotic forcing functions as Niiranen *et al.*, (2013), exc. salinity forcing on *Pseudocalanus* spp., as this forcing had negligible effects on model performance in their study.

Table 1.5: Time series forcing used in Ecosim.

Forcing series	Group(s)	Target variable	Source
Fishing mortalities (yield/biomass)	all fish stanzas	F	Cod, flounder: calculated using catch composition and fishing effort data from STECF and ICES (Bauer *et al. in prep.*). Herring and sprat: catch/biomass of the relevant age classes (Table 1.1) based on assessments (WGBFAS).
primary production forcing	phytoplankton	asymptote of growth equation	Model hindcast of the BAltic sea Long-Term large-Scale Eutrophication Model (BALTSEM, Gustafsson 2003): area-weighted average yearly P/B values, SD 25-29, excl. Gulf of Riga
Spring upper water temperature	*Acartia* spp., *Temora* spp.	Search rate	BALTSEM: average T values March-May of Gotland Sea and Bornholm Basin in depths 10-40 m
Hypoxic area (reversed)	Other macrobenthos, mysids	Search rate	BALTSEM: total hypoxic area in Gotland Sea and Bornholm Basin, yearly average
Cod reproductive volume	Cod	Egg production	BALTSEM: total cod reproductive volume in Gotland Sea and Bornholm Basin, average May-August
Summer sea surface temperature	Herring, Sprat	Egg production	BALTSEM: average T values August of Gotland Sea and Bornholm Basin in depths 0-10 m
Grey seal biomass	Grey seals	Biomass	BALSAM Grey Seal Database

There are a few additional parameters required by Ecosim compared to Ecopath. 'Feeding time adjustment rates' were set to 0.5 for vertebrates and 0 for all other groups. 'Fraction of other mortality sensitive to changes in feeding time' was set to 0 for top predators (seals, birds and adult cod). The model selection procedure for Ecosim models involves finding values for the so-called 'vulnerability' parameters that result in the best fit to observed data (Table 1.6) which in our case included time series of biomass indices from surveys (all fish and zooplankton groups and phytoplankton), catches (cod and clupeids) or landings (flounder). Fit is measured as weighted sum of squared deviations (SS). In case of biomass reference time series, only relative changes are taken into account, not absolute values, thus, it is possible to use biomass survey indices without the need for rescaling. Time series are weighted differently when calculating sum of squares according to data reliability and relevance.

Table 1.6: Time series used for fitting in Ecosim, 2004-2013. Biomass and catch data refers to the same stocks as in Table 1.1 and when converting catches to units of t/km^2 we used the same areas as indicated there.

Group name	Time series type	Weight	Source
Adult cod	Biomass- survey	80	BITS Q1 CPUE of fish >=33 cm, numbers multiplied by average weight, average of SD 25-29.
	Catches	100	WGBFAS, Catch in numbers (incl. Misreporting correction and discards) * WECA; Age3+
Juvenile cod	Biomass- survey	1	BITS Q1 CPUE of fish <33 cm, numbers multiplied by average weight, average of SD 25-29.
	Catches	10	WGBFAS, Catch in numbers (incl. Misreporting correction and discards) * WECA; Age1-2
Adult herring	Biomass- survey	80	BIAS Q4 survey indices by age, numbers * average weight; Age2+
	Catches	100	landings data from WGBFAS; Age2+; Gulf of Riga excl.
Juvenile herring	Biomass- survey	1	BIAS Q4 survey indices by age, numbers * average weight; Age1
	Catches	10	landings data from WGBFAS; Age1; Gulf of Riga excl.
Adult sprat	Biomass- survey	80	BIAS Q4 survey indices by age, numbers * average weight; Age2+
	Catches	100	landings data from WGBFAS, Age2+
Juvenile sprat	Biomass- survey	1	BIAS Q4 survey indices by age, numbers * average weight; Age1
	Catches	10	landings data from WGBFAS, Age1
Adult flounder	Biomass- survey	50	WGBFAS, weighted (by landings) mean of rescaled survey indices from SDs 24-25, 26-28

			(fish >= 20 cm) and 27 (stations Muskö, Kvädöfjärden).
	Catches	60	WGBFAS, total landings from SDs 25-29.
Acartia spp., *Temora* spp., *Pseudocalanus* spp., Other zooplankton	Biomass-survey	1	NMFRI data (station P40/P140, located in ICES rectangle 40G86, yearly average of months 8-9), described in D2.4
Phytoplankton	Biomass-survey	1	NMFRI, average value of open water stations

During the model selection process, first we assessed the sensitivity of SS to the number of 'vulnerability blocks' (v-s) fitted using the 'Stepwise fitting' plug-in of Ecosim. The plug-in iteratively fits the model, changing an increasing number of v-s compared to the default value and calculates SS in each iteration. SS decreased when fitting additional v-s until appr. 24 v-s fitted.

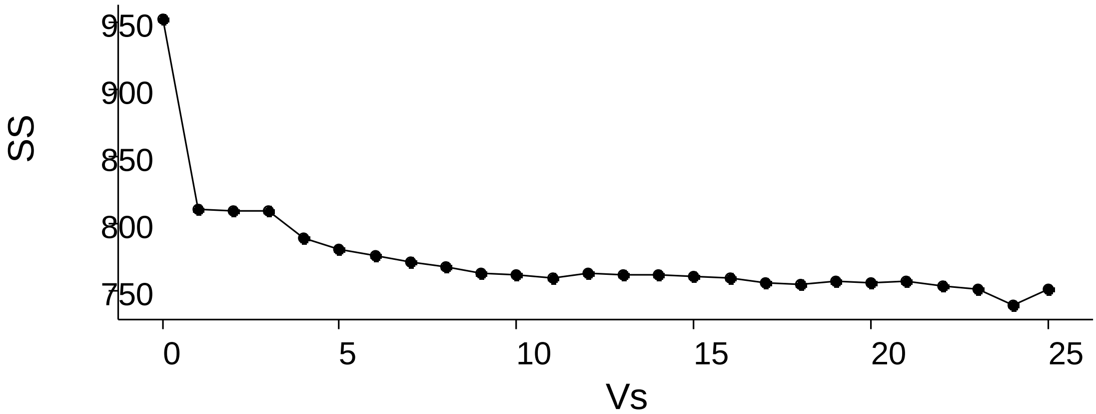


Figure 1.2 Change in SS as a function of number of v blocks fitted, calculated by the Stepwise fitting plug-in.

The plug-in doesn't include forcing functions, but during earlier tests it proved reliable in indicating the number of v-s to fit. Afterwards, we searched for the 24 v-s that affected the model fit the most using the 'Fit to time series' tool, then we searched for the best values for these v-s using the same tool, but excluding those predator groups we had no time series data for (3 groups). The model fit measured by SS and assessed visually did not differ substantially between including or excluding these groups. Thus, to only use the minimal number of free parameters necessary and avoid fitting v-s for predators we have no reference data for, we used a model fitted using 21 v-s.

We further modified v-s of adult and juvenile cod and sprat, as with fitted v-s the 'MSY Search' procedure produced unrealistically high catches at high F-s for these groups, as is often the case in Ecosim models when using relatively low or default v-s (Heymans *et al.*, 2016). This change did not largely influence model fit. As described for the 2015 North Sea key run (ICES, 2015b), we obtained an initial estimate for v using the formula v_init = [1-(Bunf/Bo)]/[1-(e/M)(Qo/Bo)], where, e is the growth efficiency (P/Q), Bunf is historical max biomass, Bo is biomass in model base year, M is the base total natural mortality rate for the predator, and Qo/Bo is the ecopath base Q/B for the species. This gave a value of 0.52 for adult cod, 14.63 for juvenile cod, 5.06 for adult sprat and 41.65 for juvenile sprat. Again, to minimize the number of free parameters, we chose to use the same value for both stanzas (10 for cod and 20 for sprat). Final SS was 787.2. Final model fit is shown on Fig. 1.3, Fig. 1.4 and Fig. 1.5.

Besides model fit to data, we applied additional tests to check for reasonable model behaviour. Successful stability testing meant that when not applying any forcing functions and keeping fishing at Ecopath level, all modelled biomasses were stable, and when stopping fishing for a short period only, biomasses returned to stability after continuing fishing again. We also tested the emerging stock-recruitment relationship in the model by applying a large range of fishing mortalities (0- 70x Ecopath level) simultaneously on all species to have a large range of adult biomasses (Figure 6). As mentioned before, we also used the 'MSY Search' tool to investigate the relationship between F and catches. This investigation was performed two ways. During the 'stationary' assessment Ecosim runs a long-term simulation at all levels of F and only the biomass of the targeted stanza reacts to changes in fishing mortality but the biomasses of all other species are kept constant. Thus, this analysis does not take into account indirect effects on the biomass of the targeted stanza via trophic linkages. In contrast, during the 'non-stationary' assessment indirect effects are taken into account to some extent. As a decrease in predator biomass at high F-s in general results in increasing prey biomasses, thus, more available food for the predators, full compensation assessments usually result in higher F values resulting in the highest catch than stationary assessments (Table 1.7).

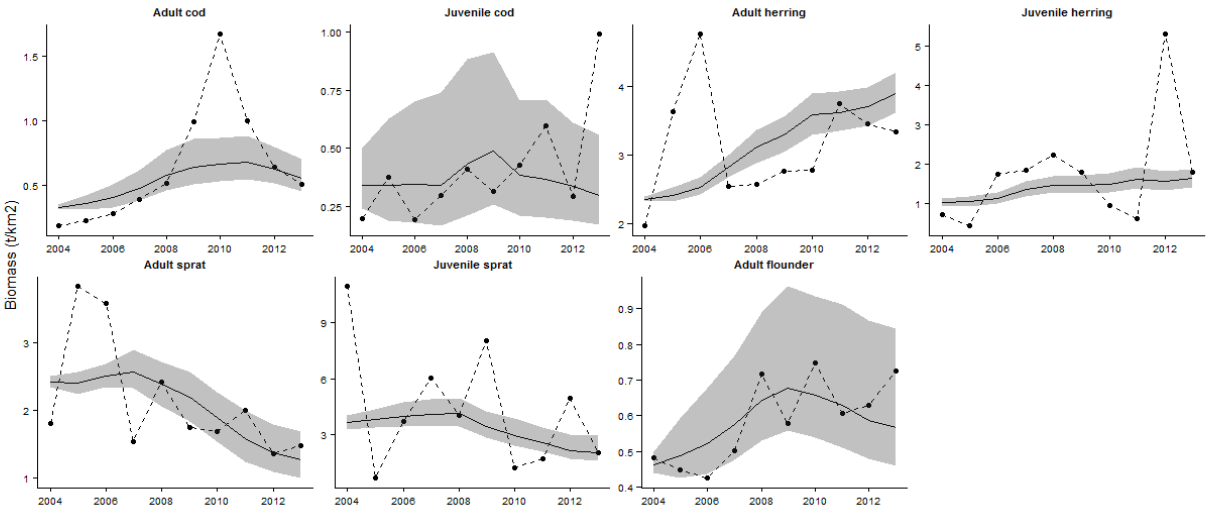


Figure 1.3 Ecosim predicted biomasses of higher trophic level species (solid lines) fit to time series of surveys (black circles, dashed lines) after final calibration of vulnerabilities. Fish survey indices of abundance (BITS, BIAS, ICESsurv) are rescaled for visualization. Grey areas indicate 10% confidence intervals based on Monte Carlo simulations varying Ecopath input biomasses, P/B and Q/B parameters and ecotrophic efficiency of mysids.

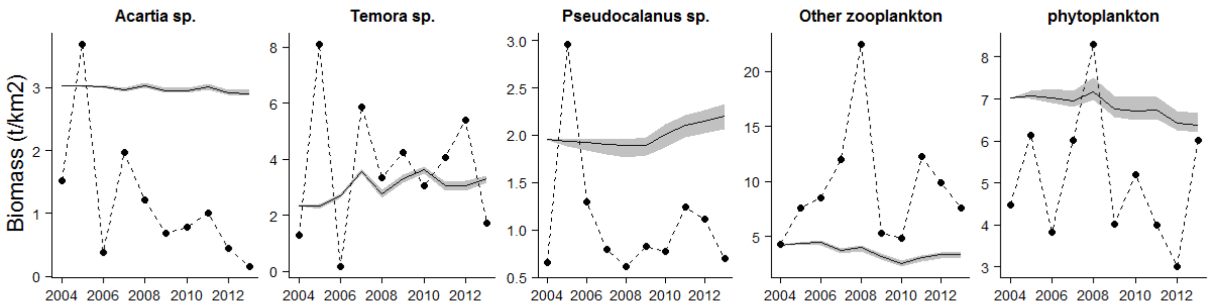


Figure 1.4 Ecosim predicted biomasses of lower trophic level species (solid lines) fit to time series of surveys (black circles, dashed lines) after final calibration of vulnerabilities. Grey areas indicate 10% confidence intervals based on Monte Carlo simulations varying Ecopath input biomasses, P/B and Q/B parameters, biomass accumulation parameter of grey seals and ecotrophic efficiency of mysids.

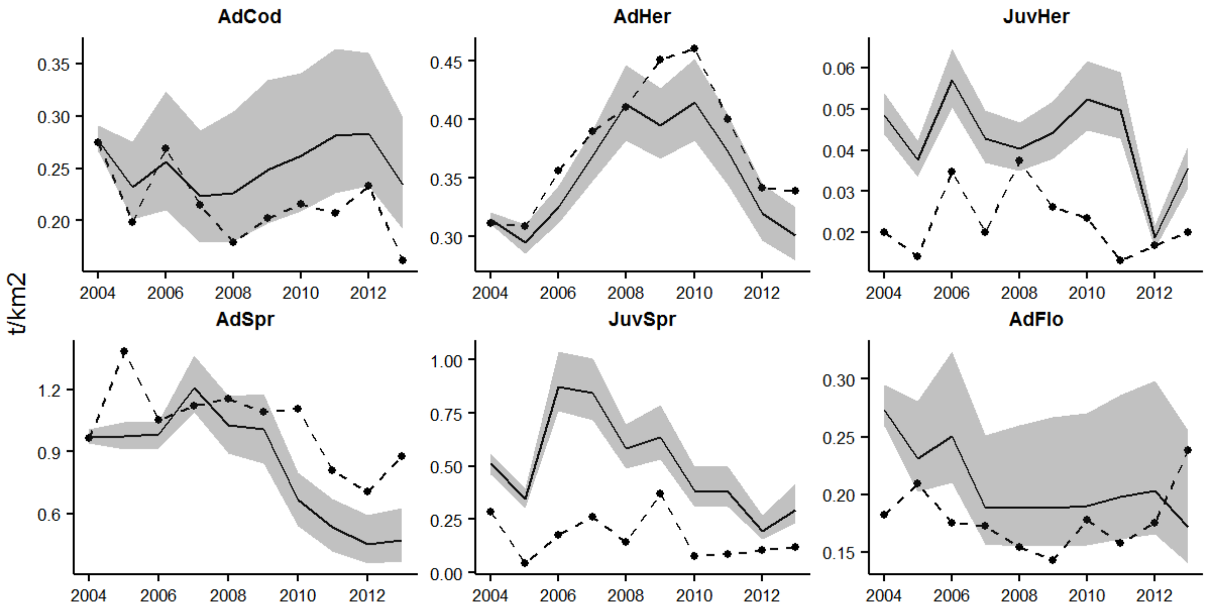


Figure 1.5 Ecosim predicted catches (solid lines) fit to time series of catches/landings for flounder (black circles, dashed lines) after final calibration of vulnerabilities. Grey areas indicate 10% confidence intervals based on Monte Carlo simulations varying Ecopath input biomasses, P/B and Q/B parameters and ecotrophic efficiency of mysids.

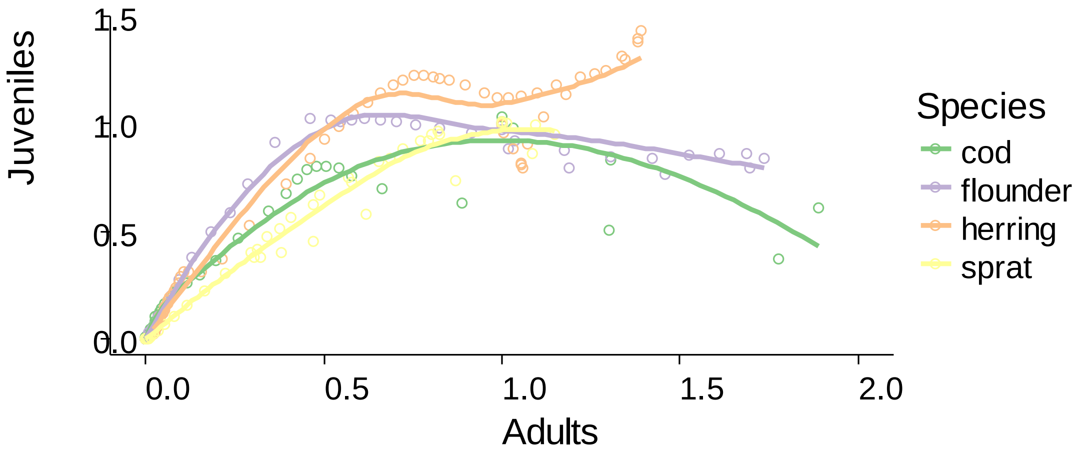


Figure 1.6 Emergent SR relationship from the model: number of adults vs. number of juveniles. Individual dots are simulated years, lines are a smooth fit for better visualization, generated by geom_smooth in the R package ggplot2.

Table 1.7 F's resulting in highest catch at equilibrium ('FMSY') predicted by EwE compared to values reported in ICES (2016b).

Group	F_{MSY} stationary	F_{MSY} full compensation	F_{MSY} assessment	Other
JuvCod	0.26	0.25		
AdCod_3	0.59	1.1	At present not defined, last accepted value (ICES, 2013) 0.46.	At present not defined, last accepted value (ICES, 2013): Multispecies FMSY (SMS) 0.55
JuvHer	0.1	0.1		
AdHer_2	0.4	0.48	0.22	MSY Flower-upper (AR): 0.16-0.28
JuvSpr	0.34	0.38		
AdSpr_2	0.59	0.83	0.26	MSY Flower-upper (AR): 0.19-0.27
JuvFlo	0.1	0.1		
AdFlo_3	0.68	0.77		

Gadget

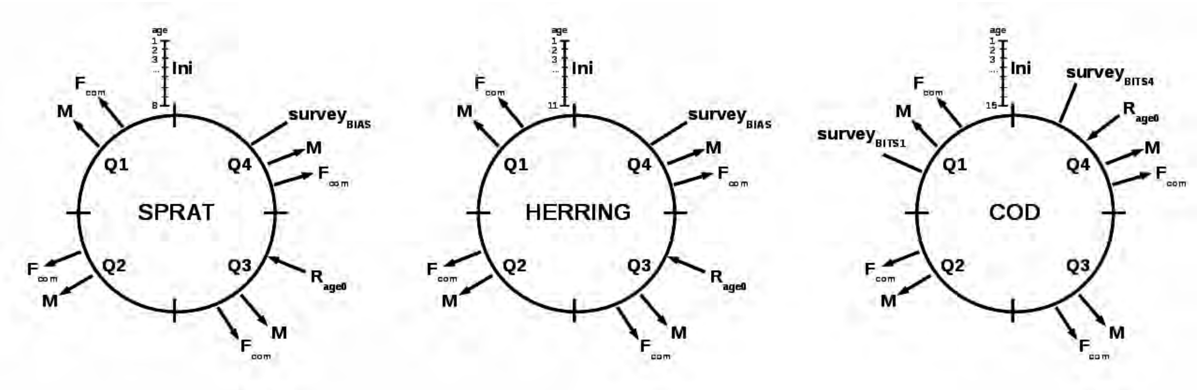
MareFrame offerred the opportunity to apply for the first time a Gadget modelling framework to cod, sprat and herring in the central Baltic Sea ecosystem. The implementation of this type of Gadget multi-species models requires a number of intermediate steps, including the parametrization of single species 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 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)).

Conceptual models

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.



All the three models cover the same time period

Single species sprat

Biological model

The modelled sprat population spans from age0 to age10+ and from 3.5 to 17.5 cm in length with 1 cm length resolution. Natural mortality at age is expressed by the following vector: 0.5, 0.44, 0.44, 0.44, 0.43, 0.42, 0.42, 0.41, 0.41, 0.41, 0.41 calculated as average of the natural mortality matrix used in WGBFAS assessment which is derived from the MSVPA. This is a one area one stock model with no implementation of sex and sub-population structures. In practice, no difference in any biological parameter is assumed between males and females, or among different stages such as juvenile and adults.

Recruitment is estimated to occur once a year during quarter 3. At this timestep age0 fish enter the model with a fixed mean length (estimated in the model) and standard deviation (estimated during preliminary runs) of 7.7 $\pm$ 0.9 cm. No stock-recruitment relationship is used to estimate the number

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of recruits. A year-specfic recruitment parameter is estimated for each year of the model. As initial values of recruitment parameters we used assessment estimates of recruitment from WGBFAS. Sensitivity analysis showed some influence of the initial value of parameters on the final estimation.

Mean growth is implemented with a simple von Bertalanffy model where the increase in length for each length group i and the corresponding increase in weight are given by the following two equations:

$$dL_i = (L_{inf} - L_i)\,(1 - \exp(-k\,dt)) \qquad (1)$$

$$dW_i = a\,((L_i + dL_i)^b - L_i^b) \qquad (2)$$

where:

< dt > is the length of the timestep

< L_{inf} > is the asymptotic length at which growth is zero

< k > is the growth rate

< a > is the linear coefficient of the length-weight model

< b > is the exponential coefficient of the length-weight model

Dispersion around the mean growth in length is implemented with a beta-binomial distribution. Such implementation has the flexibility to produce non-symmetrical distributions with larger right-end tail as the curve dispersion increases (see Taylor et al. (2007) and Begley (2006) for more details).

Individual weight-at-age reported in the commercial catches shows a mark change during the study period, with high weights in the 1970s-1980s and low weights from the end of the 1990s. To capture this strong signal from the biological data, the length-weight relationship was assumed to change over three time blocks represented by the periods 1974-1989, 1990-1996, 1997-2013, and the associated parameters were estimated during preliminary Gadget runs.

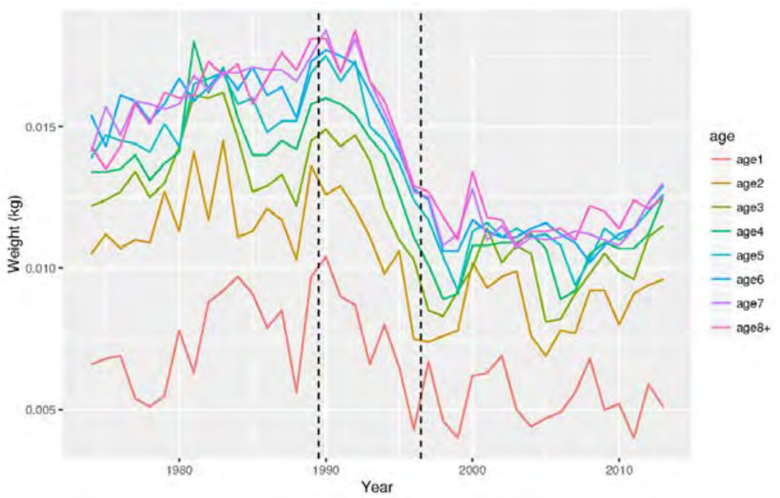

Figure 1.7. Observed weight-at-age of sprat from commercial catches in 1974-2013

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The process of maturation is not implemented in the current model. The same constant vector with the proportion of mature fish at age (age0-3+: 0, 0.17, 0.93, 1) is used to calculate the spawning stock biomass from the estimated number of fish and mean weight-at-age.

Fleets

The model includes one commercial fleet and one survey (BIAS). A sigmoid function defined by two parameters is assumed for the suitability (i.e., combination of selectivity and availability) of both fishery and survey:

$$S(l) = 1 / (1 + \exp(-a -b\, l) \qquad (3)$$

where:

< a/b > is the length at which 50% of fish is selected by the gear

< b > is the parameter influencing the steepness

Catch amount as biomass of sprat extracted quarterly from the population is assumed to be exact in the model.

Likelihood components

The Baltic sprat model is parametrised using the following likelihood components:

```
* Age distribution commercial fishery
```

Number of sprat at age (age0-age8+) caught by all the commercial fisheries for each quarter during the time period 1974-2013. A sum of square likelihood function is used to compare the age distribution from the model with the age distribution from this dataset as:

$$l = \mathrm{sum}_t\ \mathrm{sum}_a\ (P_{tral} - pi_{tral}) \qquad (4)$$

where:

< P > is the proportion of the data sample for that time/age combination

< pi > is the proportion of the model sample for that time/age combination

```
* Stock and recruitment index
```

Indices of abundance for the stock and age0 fish are calculated from the BIAS acoustic survey. A similar likelihood component is used for both these indices represented by the sum of squares of the log-linear regression of the difference between the modelled index of abundance and the abundance index of all sprat or only age0 fish from the acoustic survey.

$$l = \mathrm{sum}_t\ (\log(I_t) - (a + b \log(N_t)))^2 \qquad (5)$$

where:

< I > is the observed survey index

< N > is the corresponding index calculated in the Gadget model

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The intercept (a) and slope (b) of the log-linear regression are estimated for both the indices.

```
* Length distribution survey
```

A length distribution by 1-cm length interval for the stock is derived for each year of the BIAS using data collected in the pelagic trawl samples associated to the acoustic survey. Length distributions in number for the entire stock are calculated by weighting the length distribution at ICES rectangle level by the corresponding acoustic index of abundance. A sum of square likelihood function is used to compare the length distribution from the model with the length distribution from this dataset as

$l = sum_t \ sum_l \ (P_{tral} - pi_{tral})^2$ (6)

where:

< P > is the proportion of the data sample for that time/length combination

< pi > is the proportion of the model sample for that time/length combination

```
* Age-length key
```

Number of sprat by age and length (at 1-cm length interval) is calculated for each year of the BIAS using biological samples collected in the pelagic trawl hauls associated to the acoustic survey. Similarly to the length distribution, also the contribution of the number of fish for each age-length combination at the ICES rectangle level is weighted by the corresponding acoustic index of abundance. A sum of square likelihood function is used to compare the age-length distribution from the model with the age-length distribution from this dataset as

$l = sum_t \ sum_a \ sum_l \ (P_{tral} - pi_{tral})^2$ (7)

< P > is the proportion of the data sample for that time/area/age/length combination

< pi > is the proportion of the model sample for that time/area/age/length combination

```
* Weight-at-age
```

Quarterly weight-at-age (waa) from the commercial fishery were used to estimate the length-weight parameters during preliminary Gadget runs. The likelihood function used to compare this data component to the model calculates a weighted sum of squares of the mean weights, with the weighting given by the variance of weight of the input data as follows:

$l = sum_t \ sum_a \ (((x_{ta} - mu_{ta})^2 / s_{ta}^2) \ N_{ta})$ (8)

where:

< x > is the mean weight-at-age from the data

< mu > is the mean weight-at-age calculated from the model

< s > is the standard deviation of the weight-at-age from the data

< N > is the number-at-age in the commercial fishery

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Fitting

A genuine estimation is achieved for all the parameters except the β parameter of the beta-binomial implementation of growth and the last age group of the initial population which were estimated at the upper and lower bound, respectively. Alternative fixed values of β far from the boundaries have been tested and not major impact was observed on the model output, suggesting that the issue is of minor relevance and with more time it could be likely addressed within the estimation procedure. Further investigations are required for the last age group of the initial population, but it is expected to be of minor relevance importance.

The likelihood scores gained by the survey related data components are smaller than the score for the age distribution from the commercial fleet. Only exception is the recruiment index provided by the survey.

The modelled age distributions for the commercial fleet show a high level of agreement with the observations also at a quarterly timestep. Fitting is generally poorer on the age distributions from the first few years. Strong cohorts are clearly visible from the data and generally well represented by the model (i.e., high recruitment of of 1997).

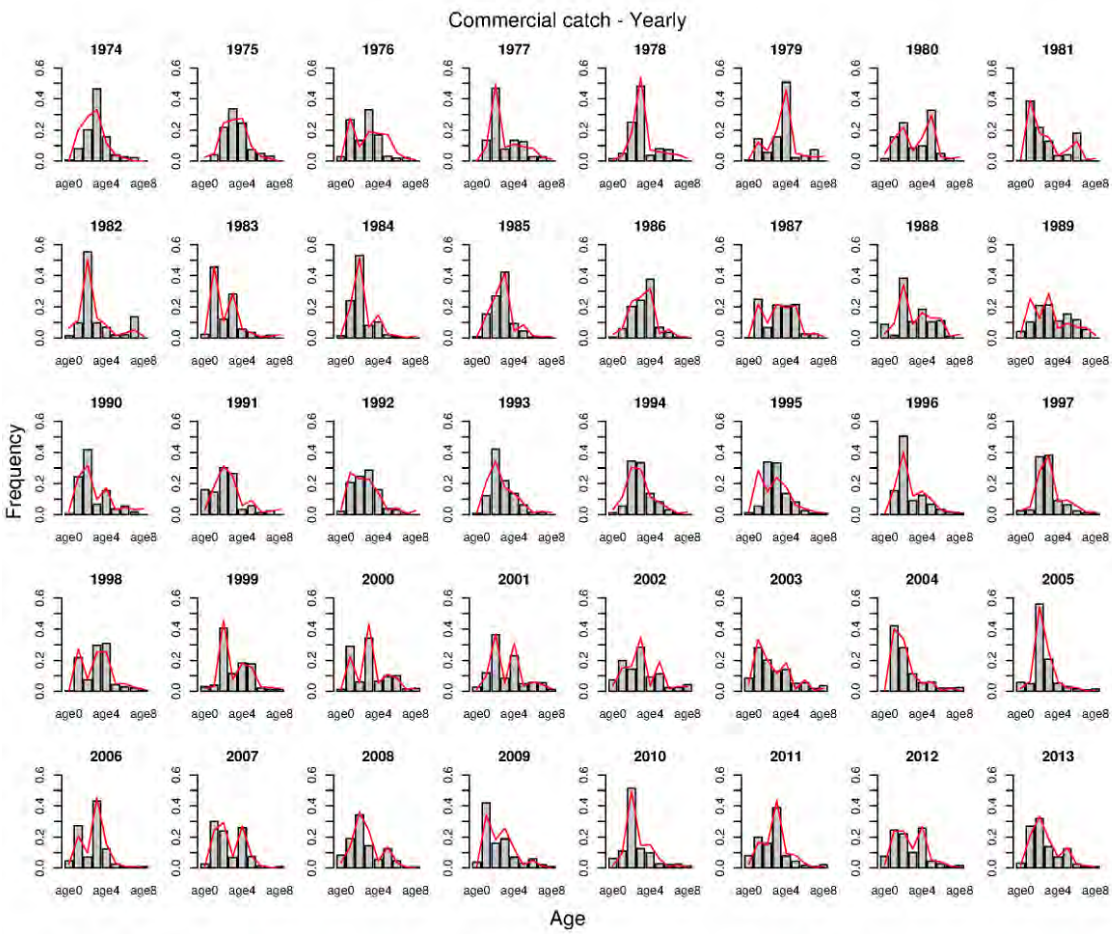

Figure 1.8. Observed (bars) and predicted (red line) annual average age distribution from commercial catches of sprat in 1974-2013.

The L_{inf} of 13.4 cm (estimated outside the model) and k of 0.458 result in a good average fitting of the age-length key. Growth in sprat approaches quickly the estimated asymptotic length after approximately 5-6 years.

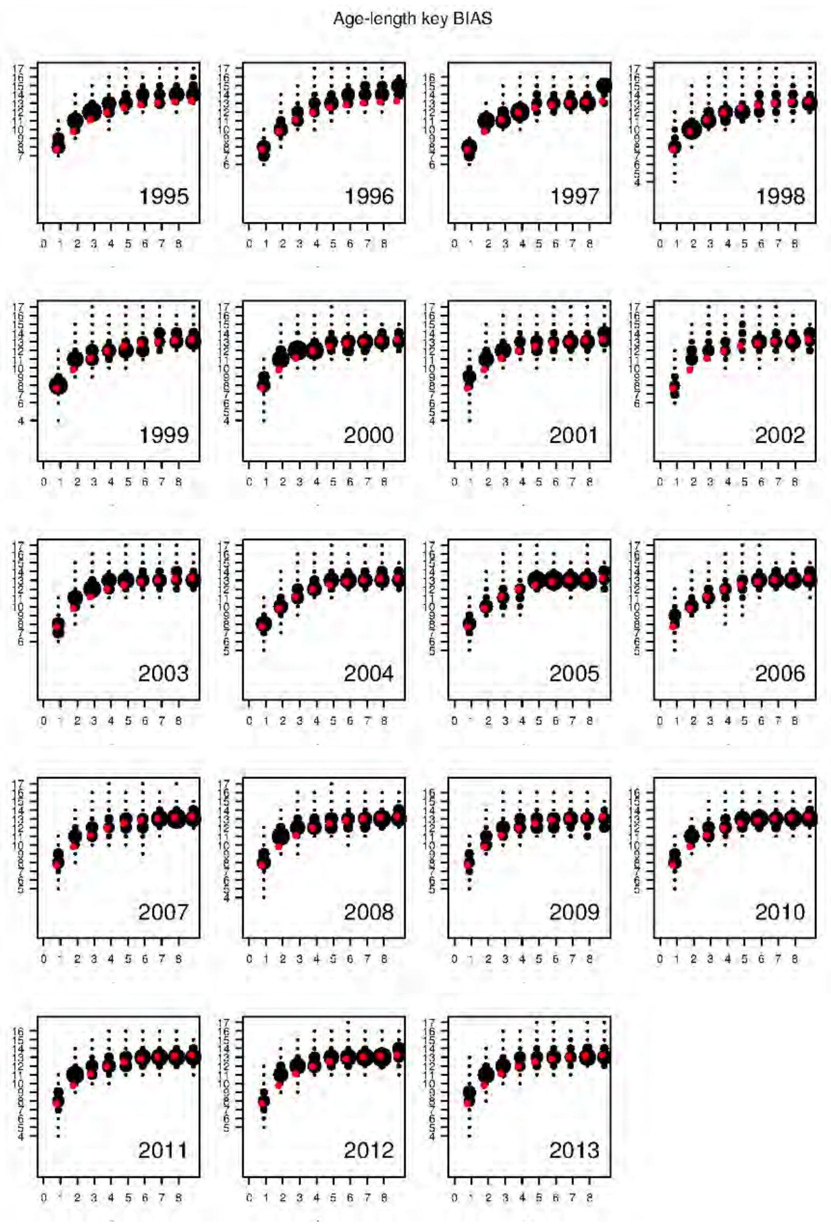

Figure 1.9. Observed (black bubbles) and predicted (red dots) age-length distribution of sprat from BIAS survey in 1995-2013.

The length distribution from the survey is quite irregular along the time series showing a first distinct peak of small sprat aroud 8 cm only in some years (1997, 2002, 2003, 2008, 2010-2013). The recruitment (r^2=0.65) and especially the abundance (r^2=0.91) index present both a large agreement with the observations. Years 1995 and 1997 show the largest residuals in the recruitment index with observations 3 times smaller than predictions. The estimated spawning stock biomass (SSB) shows good agreement with output from a SAM model (downloaded from) which is also in agreement with the current assessment of the stock. After a period of small stock size throughout the 1970s and 1990s, the sprat stock peaked around the mid 1990s to decrease again to intermediate levels throughout the 2000s.

Single species herring

The single-area and single stock model for central Baltic herring spans from age0 to age15+ and from 4.5 to 35.5 cm in length with 1 cm length resolution. Natural mortality at age follows the following vector: 0.5, 0.31, 0.29, 0.27, 0.26, 0.25, 0.24, 0.23, 0.23, 0.23.

Age0 recruits with mean length 10 cm and standard deviation of 2.5 enter the model once a year in quarter 3. Similarly to sprat, recruitment estimates of recruitent were used as initial parameters and recruitment was then estimated in Gadget together with other parameters.

Von Bertalanffy growth model was adopted to determine the increase in length for each length group (eq. 1), where the parameter k was estimated while Linf fixed to 24.7 cm. The beta parameter of the beta-binomial distribution around growth was estimated.

Inspection of weight-at-age from commercial data shows marked changes in the condition of herring, with an overall decrease in weight which is more pronounced in the end of the 1970s and 1980s and in the older age groups. For the some age groups the observed decrease in weight was >60% over the time period of interest for the model. Seasonal differences characterise this general pattern with a progressive decrease in quarter 1 and 2, while weights are relatively stable and drop only at the end of the 1980s and early 1990s during quarter 3 and 4. If the general decrease may be explained in terms of limiting resources as a consequence of trophic interaction (ie, with sprat) and a changing environment, seasonal differences in the pattern are somehow more difficult to explain. Among the possible reasons we may include an underlying diversity in the spatial distribution of condition of central Baltic herring which could emerge as a seasonal pattern if the contribution to the overall catch of the different countries operating in different areas of the Baltic also changes between quarters. Another possible explanation may be the mixing between central Baltic herring and other Baltic herring stocks (ie, western Baltic, Gulf of Riga, Bothnian Sea).

To represent the overall change in weight, the increase in weight as a function of length (eq. 2) was implemented over the following three time blocks: 1974-1985, 1986-1995, 1996-2013.

Fleets

Herring is harvested in the model by one commercial fleet corresponding to the pelagic fishery. Suitability is defined with a sigmoid functional form with parameters estimated in the model. Catches expressed as biomass of herring are extracted quarterly from the modelled population.

Similarly to sprat, the BIAS survey is used for the herring model.

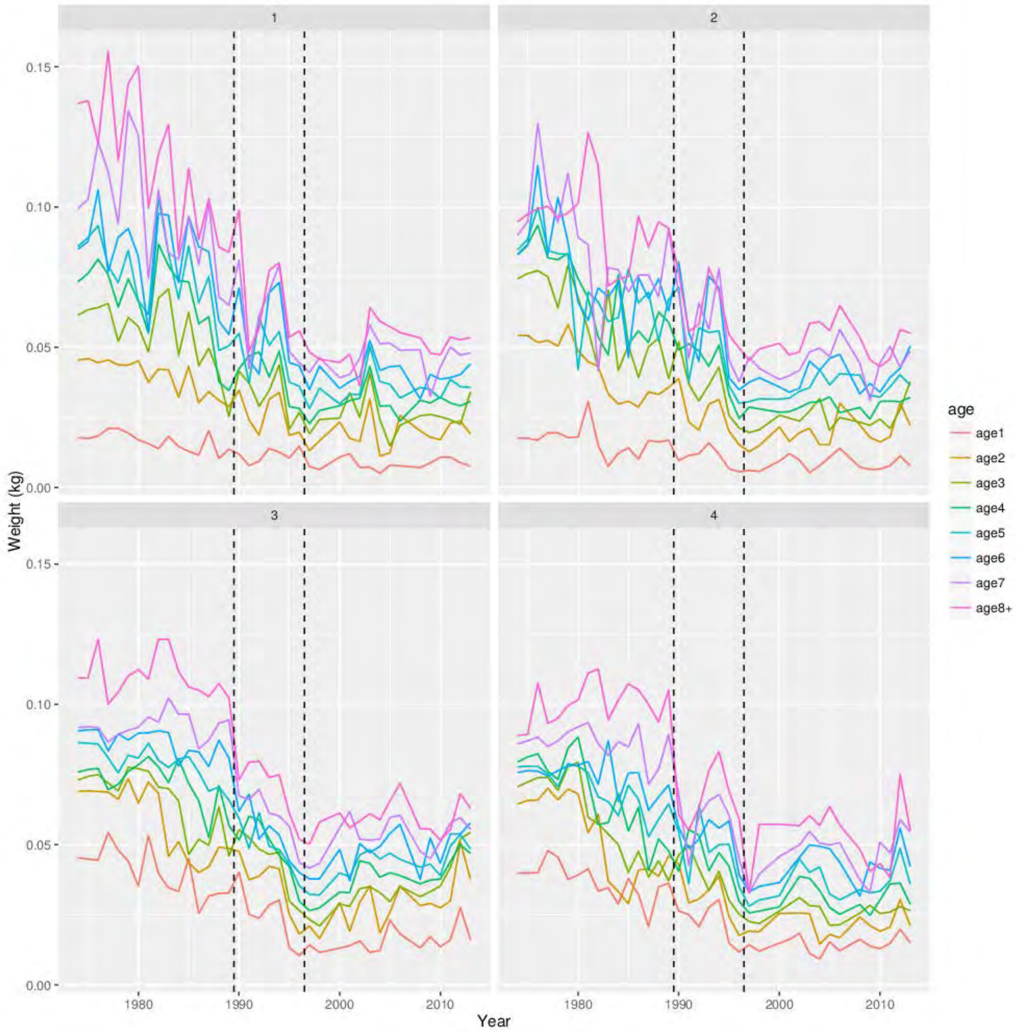

Figure 1.10. Observed weight-at-age of herring by quarter from commercial catches in 1974-2013

Likelihood components

Herring and sprat dominate the same pelagic environment in the Baltic and they mostly occur as a mixed catch in the same commercial fisheries and scientific surveys. The two species are not only sampled by the same acoustic survey (BIAS), but also occur in the same associated pelagic trawl samples suggesting that the two species form also mixed schools. Although the dependence of herring and sprat catches and the mixed nature of the associated samples, for simplification the data and likelihood for the two species are treated independetly in the present single and multispecies Gadget implementations. Several of the likelihood functions used for the sprat model are also adopted in a similar way on the herring data.

```
* Age distribution commercial fishery
```

Number of herring at age (age0-age8+) caught by all the commercial fisheries for each quarter during the time period 1974-2013.

```
* Length distribution survey
```

Length distributions of herring by 1-cm length interval are calcuated for each year of the BIAS survey.

```
* Age-length key
```

Number of herring by age and length (at 1-cm length interval) is calculated from the biological samples of the BIAS survey.

```
* Survey indices by age
```

Survey indices by age for age0 to age8+ are available from BIAS survey. A likelihood function similar to eq. 5 is used to fit the time series for each age group.

```
* Weight-at-age
```

A 'catchstatistics' Gadget likelihood function was used to fit quarterly weight-at-age data from the commercial fishery and from the BIAS survey according to the eq. 8.

Fitting

The model shows an overall good fitting of the main datasets, with the only exception of the weight-at-age likelihood components. The indices of abundance by age appear rather noisy and characterise by possible outliers (ie, in year 2000 age5+) but the model is able to capture their main pattern.

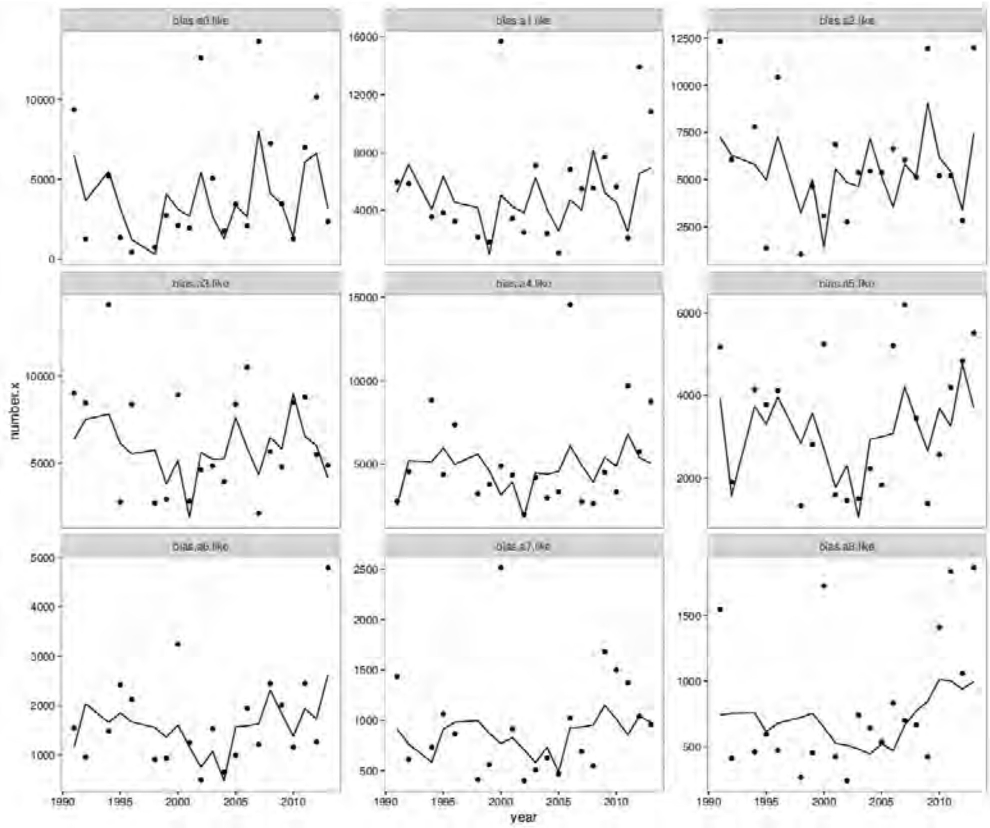

Figure 1.11. Observed (dots) and predicted (lines) indices of abundance by age of herring from BIAS survey in 1991-2013.

Commercial age distributions are well represented by the model which in most years is able to represent the most abundant age groups and individual peaks in the distribution which are highly variable from one year to another and multimodal in some cases (ie, 1988-1992).

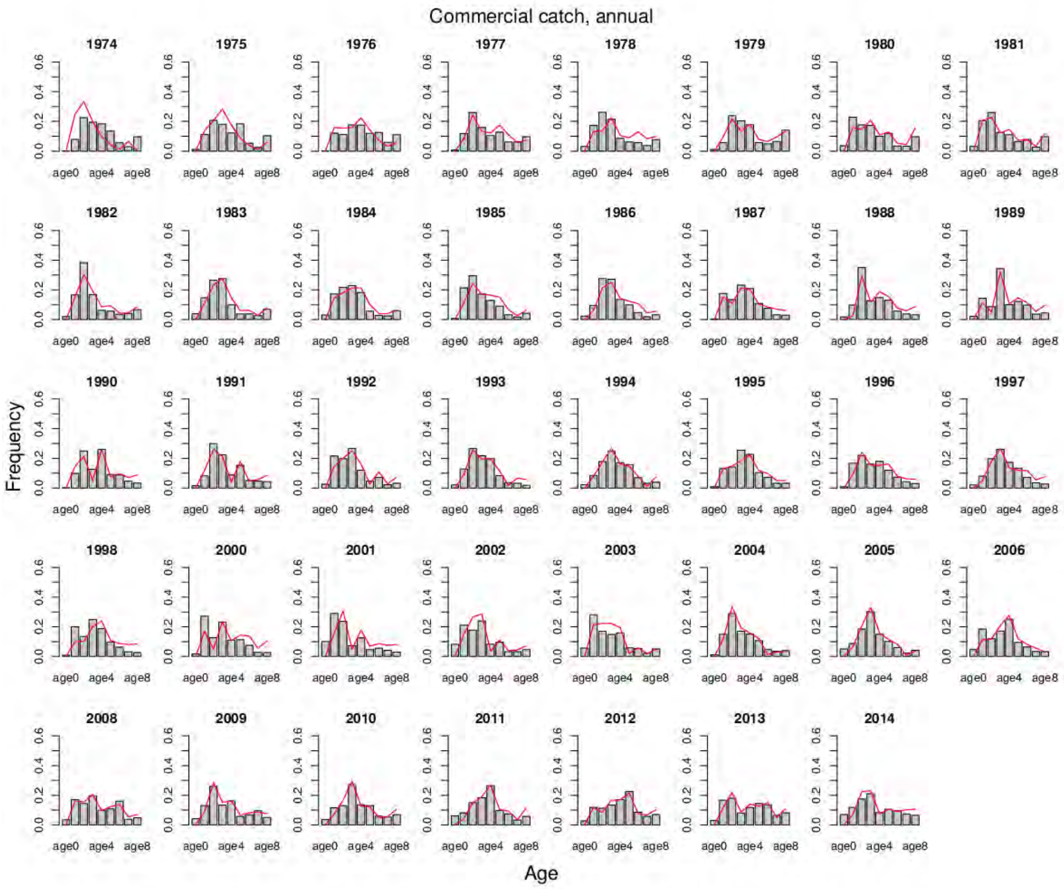

Figure 1.12. Observed (bars) and predicted (red line) annual average age distribution from commercial catches of herring in 1974-2013.

Length-weight parameters estimated for the three time blocks represent a progressive reduction in the herring weight, but the reduction is too moderate to capture the extent of the decrease. The very low weight assigned by the model to both the weight-at-age likelihood components is consistent with the poor fitting of this component. A possible explanations may lay in the occurrence of additional processes other than changes in the length-weight relationship, such as changes in growth, that the model is not able to represent and that may contribute to determine the observed pattern in these two likelihood components.

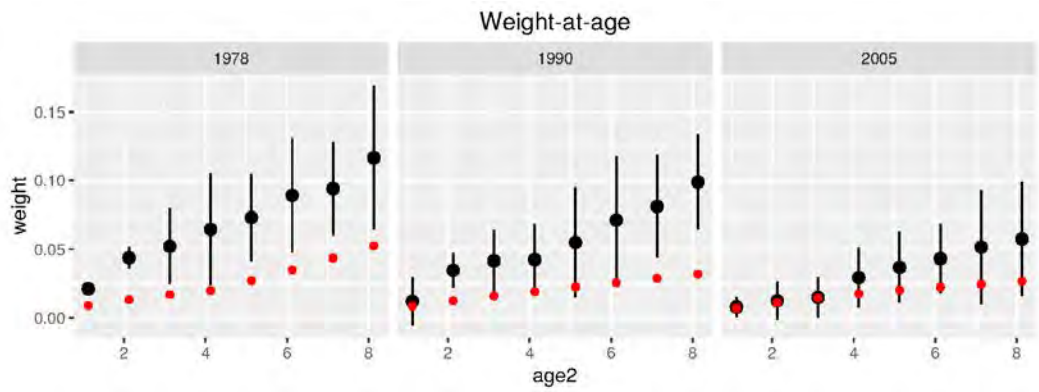

Figure 1.13. Observed (bars) and predicted (red dots) weight-at-age of herring in three different years, 1978, 1990, 2005.

Both recruitment and spawning stock biomass (SSB) have an overall good agreement with the pattern observed in the assessment. The decrease in SSB that characterised the modelled timeframe until the early 2000s and the following increase are slightly more pronounced in our model compared to the assessment estimates. Comparison of age1 herring which considered recruits in the assessment shows moderately higher estimates but with a very similar pattern.

Single species cod

Biological model

The modelled cod population spans from age0 to age15+ and from 1 to 137 cm in length with 2 cm length resolution. A vector of natural mortality at age (Mage) is derived from multispecies VPA (MSVPA). M0-1 are the MSVPA average values over the period 1974-2011, M2-4 are fixed to 0.3 (it was 0.4 from MSVPA) and M5-15 fixed to 0.2 as follows: 1.17 0.37 0.3 0.3 0.3 0.20 0.20 0.20 0.20 0.20 0.20 0.20 0.20 0.20 0.20 0.20 This is a one area one stock model with no implementation of sex and sub-population structures.

Recruits (age0) come into the model once a year in quarter 4 with mean length (estimated during preliminary runs) and standard deviation (fixed based on some exploration) of 9.17 cm and 3 cm, respectively. Also for cod, no stock-recruitment relationship is used to estimate the number of recruits. A year-specfic recruitment parameter is estimated for each year of the model based on the cohort development in the survey and commercial data. As initial values of recruitment parameters we used assessment estimates of recruitment from SAM runs attempted at WGBFAS and during the cod benchmark in 201x. Although the estimation of recruitment was not at the boundary, caution was needed on setting the parameter boundaries, the upper bound, as the model tended to easily fall outside realistic estimations. Sensitivity analysis showed some influence of the initial value of parameters on the final estimation.

Esploratory analysis of both lenth-at-age and weight-length relationships from the survey data (BITS) suggested large changes in the growth of cod throughout the time period investigated. Variability in the growth patterns showed a good temporal correlation that we tried to capture by modelling growth

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into three time blocks (1974-1998, 1999-2006, 2007-2013) with the blocks anticipated of one year for the length-weight relationship (1974-1997, 1998-2005, 2006-2013) based on visual inspection.

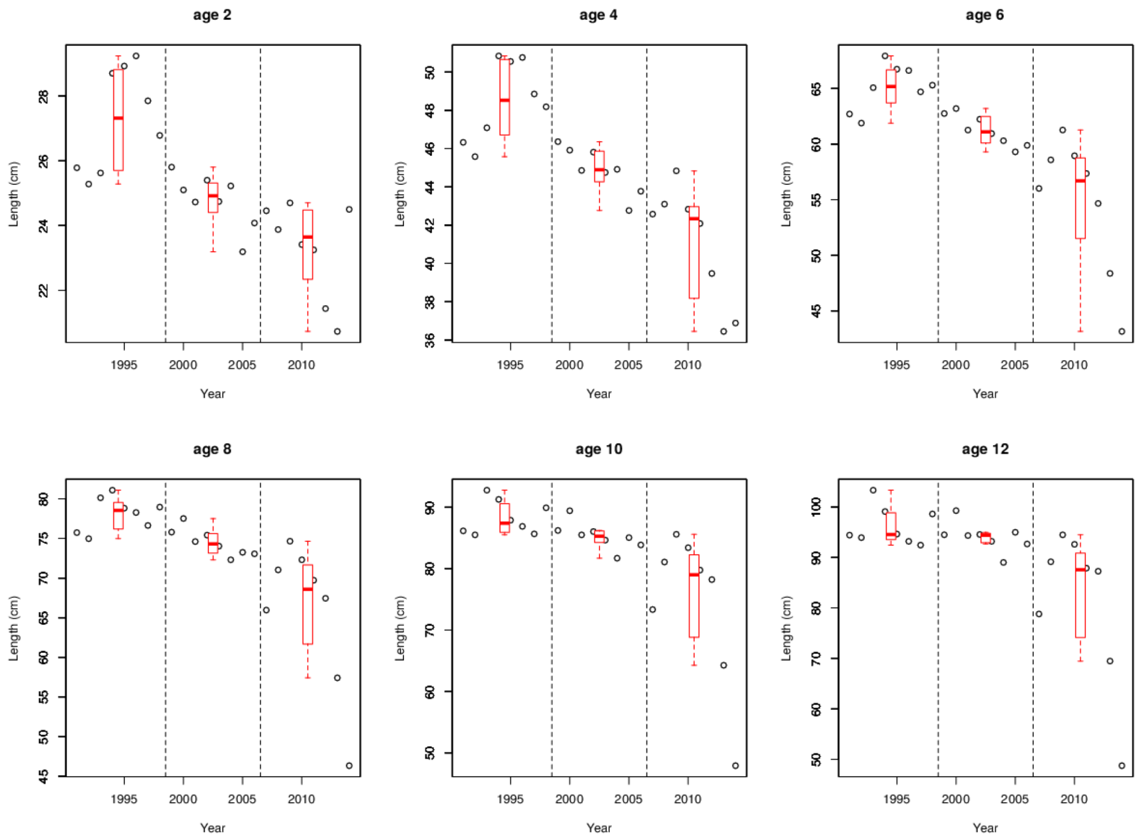

Figure 1.14. Mean length-at-age of cod from annual BITS survey data and box-plot representing the distribution of the dots in the three time periods 1974-1997, 1998-2005, 2006-2013.

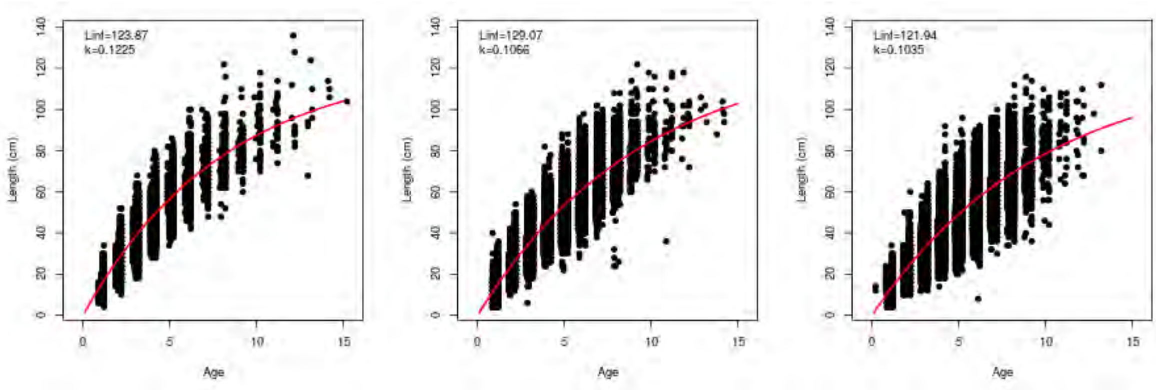

Figure 1.15. Individual length-at-age observed in the BITS (dots) and VB growth curve (red lines) estimated for the three periods 1974-1998, 1999-2006, 2007-2013.

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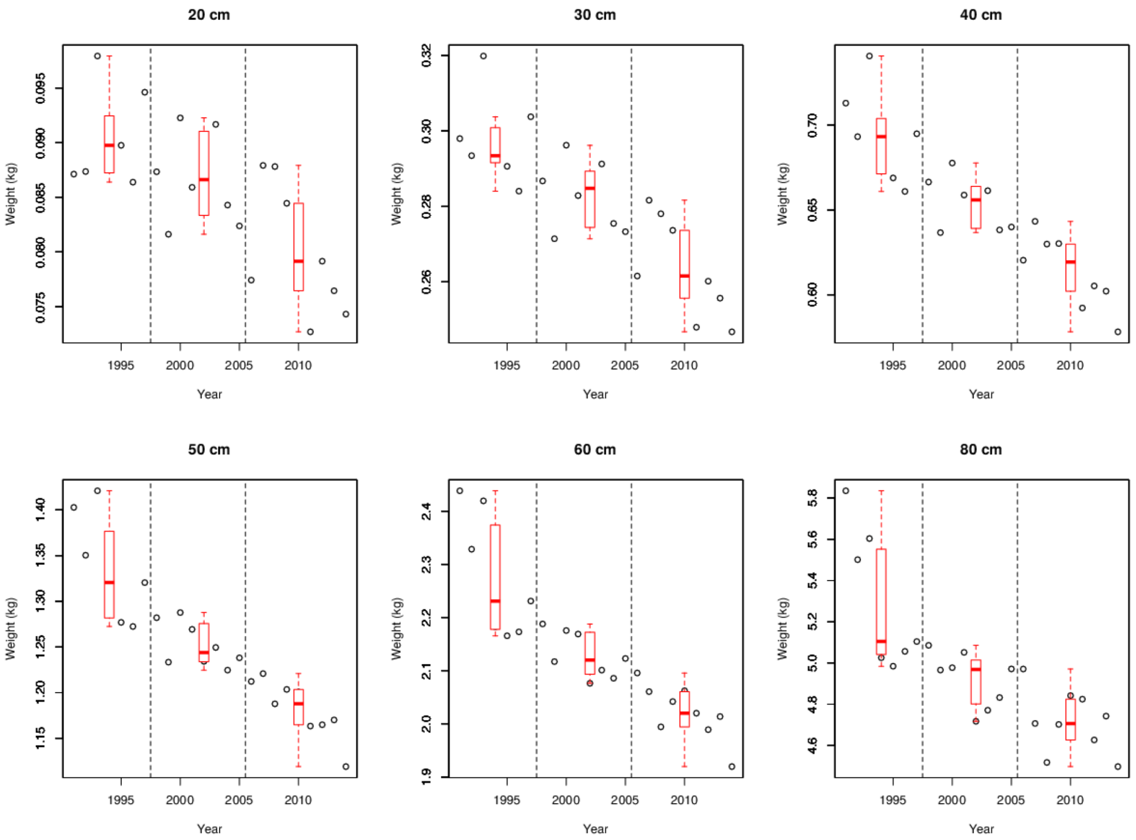

Figure 1.16. Mean weight-at-age of cod from annual BITS survey data and box-plot representing the distribution of the dots in the three time periods 1974-1997, 1998-2005, 2006-2013.

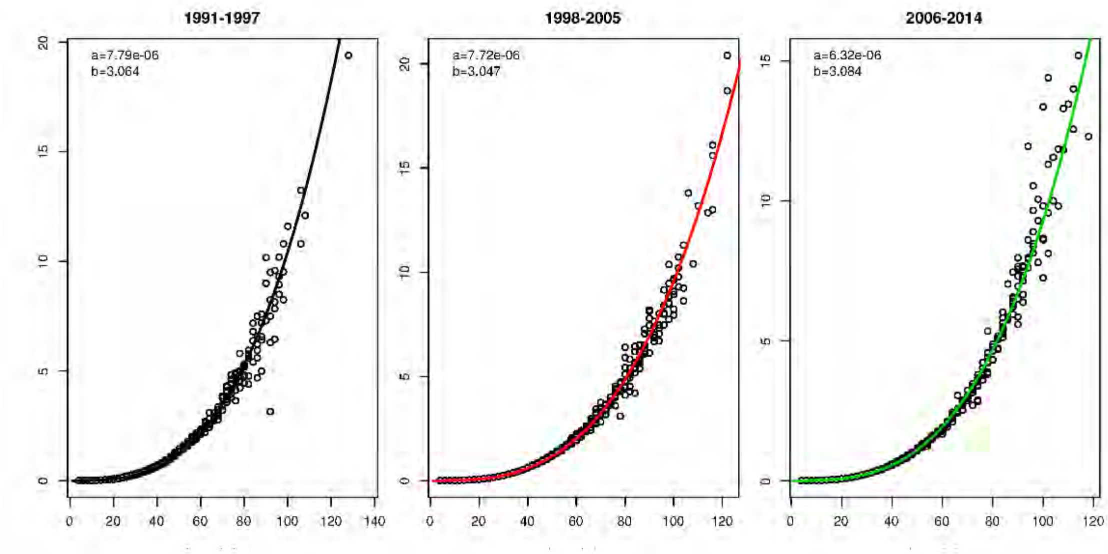

Figure 1.17. Individual weight-at-length observed in the BITS (dots) and L-W growth curves estimated for the three periods 1974-1997, 1998-2005, 2006-2013.

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The von Bertallanffy growth parameters Linf and kappa (eq. 1) and the parameters alpha and beta for the corresponding increase in weight (eq. 2) were estimated for each time block outside Gadget and fixed in the model. The parameter of the beta-binomial distribution used to implementation dispersion around the mean growth was estimated in Gadget and assumed the same for the entire time period of the model.

Maturation is not implemented in the current model and a maturity-at-age matrix from WGBFAS is used to calculate the spawning stock biomass from the estimated number of fish and mean weight-at-age.

Fleets

The cod model includes two commercial fleets corresponding to the active (ie, trawlers) and passive (ie, gillnetters) gears, and three surveys (BITS in quarter 1 during 1991-2000 [BITS11], BITS in quarter 1 during 2001-2013 [BITS12], BITS in quarter 4 during 2001-2+13 [BITS4]). Sigmoid functions (eq. 3) defined by two parameters were used to represent the suitability of all the fleets.

Catch amount as biomass of cod disaggregated by fleet and quarter were assumed to be exact for the period 1974-2003. Catches are available by gear and quarter only from 1998-onward. For the period 1991-1997 total catches were split into active and passive by quarter using the quarterly average proportion observed during the period 2000-2005, given that available information on effort suggests no major trend in the relative contribution of the two fleets during the 1990s. Prior 1991 catches were not available separated for the two gears but trawlers are known to be dominating the catch during the 1970s and 1980s in the Baltic cod fisheries (pers. comm. WGBFAS). For this reason 100% of the catches were assumed to be from the active fleet during that period. For the period 2004-2013 catches of cod were estimated in Gadget based on the available cod biomass and the fishing effort disaggregated for the two fleets:

$$C(l) = E\ dt\ S(l)\ N\ W \qquad (9)$$

where:

< E > is the fishing effort

< dt > is the length of the timestep

< N > is the number of cod in the length cell

< W > is the mean weight of cod in the length cell

< S > is the suitability function

Likelihood components

Estimation of parameters in the cod model is based on the following likelihood components:

```
* Age distribution commercial fisheries
```

Number of cod at age (age0-age8+) aggregated for the two fleets is available from each quarter. This is used to construct a likelihood component only for the active gear prior 1991 and for the combined active and passive gears for the period 1991-1999. A sum of square likelihood function is used to compare the age distribution from the model with the age distribution from these two dataset as in eq. 4.

```
* Length distribution commercial fisheries
```

Length distributions by 2-cm length interval are calculated for the active and passive fleet for each year and quarter from 2000-onward. A sum of square likelihood function is used to compare the length distribution from the model with the length distribution from this dataset as in eq. 6.

```
* Survey indices
```

Survey indices are calculated from BITS1 for the length groups <20, 20-40, 40-60, >60 cm. Length groups were defined based on visual inspection of the survey length distributions with the intent to capture the dynamics of relatively homogeneous groups of fish. For each length group a likelihood function was specified as in eq. 5. An overall index of abundance is used from BITS4 to better inform the model about changes in the stock abundance during the last decade.

```
* Length distribution survey
```

Yearly length distribution by 2-cm length interval is calculated from the BITS. A sum of square likelihood function is used to compare the length distribution from the model with the length distribution from this dataset as in eq. 5.

```
* Age-length key
```

Number of cod by age and length (at 2-cm length interval) is calculated for each year of the BITS. A sum of square likelihood function is used to compare the age-length distribution from the model with the observed age-length distributions as in eq. 7.

```
* Commercial catches
```

Estimated quarterly catches in biomass for the active and passive fleets are compared to the officially reported catches during the period 2004-2013. A sum of squared likelihood function is used for this purpose:

$$l = \text{sum}_t\,(\log(N_t) - \log(v_t))^2 \qquad (10)$$

where:

< N > is the reported catch biomass of cod for each quarter

< v > is the modelled catch biomass of cod for each quarter

Fitting

The modelled quarterly age distribution for the combined active and passive fleets shows a moderate level of agreement with the observations. Fitting is better in quarter 2 and in the second half of the 1990s. The length distribution of the two fisheries are both dominated by a single mode in all quarters and years placed around 38-40 cm for the active fleet and 44-46 cm for the passive fleet. Few exceptions are observed given by secondary peaks of small cod for the active fleet in 2008 quarter 2 and the passive fleet in 2006 and 2007 quarter 3. The model tends to slightly over estimate larger fish during the last two years of the time series, but overall captures correctly the different selectivity of the two fisheries as suggested by the overall good fitting of the length distribution of the catch in most years and quarters.

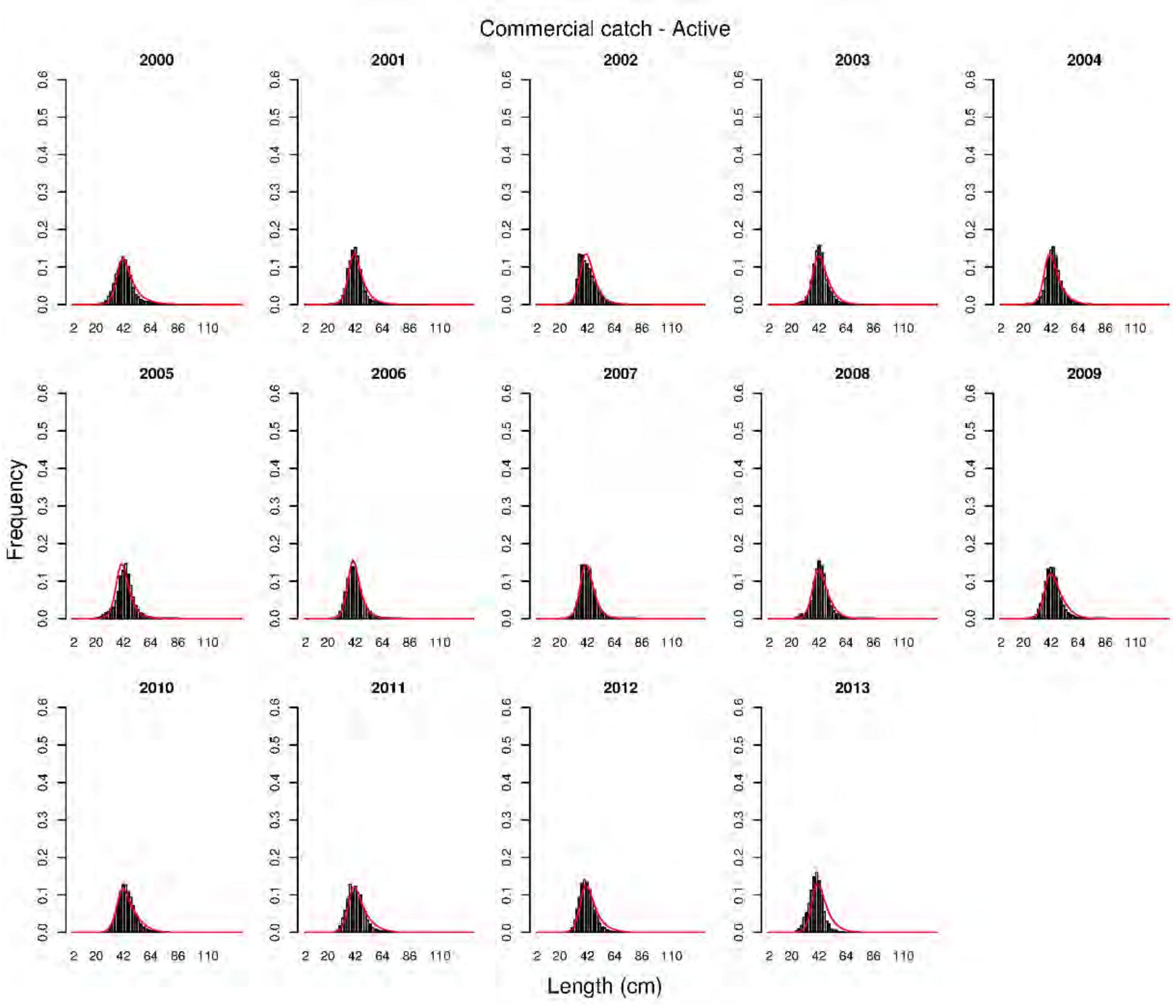

Figure 1.18. Observed (bars) and predicted (red line) annual average length distribution of cod from the commercial catches of active gears in 2000-2013.

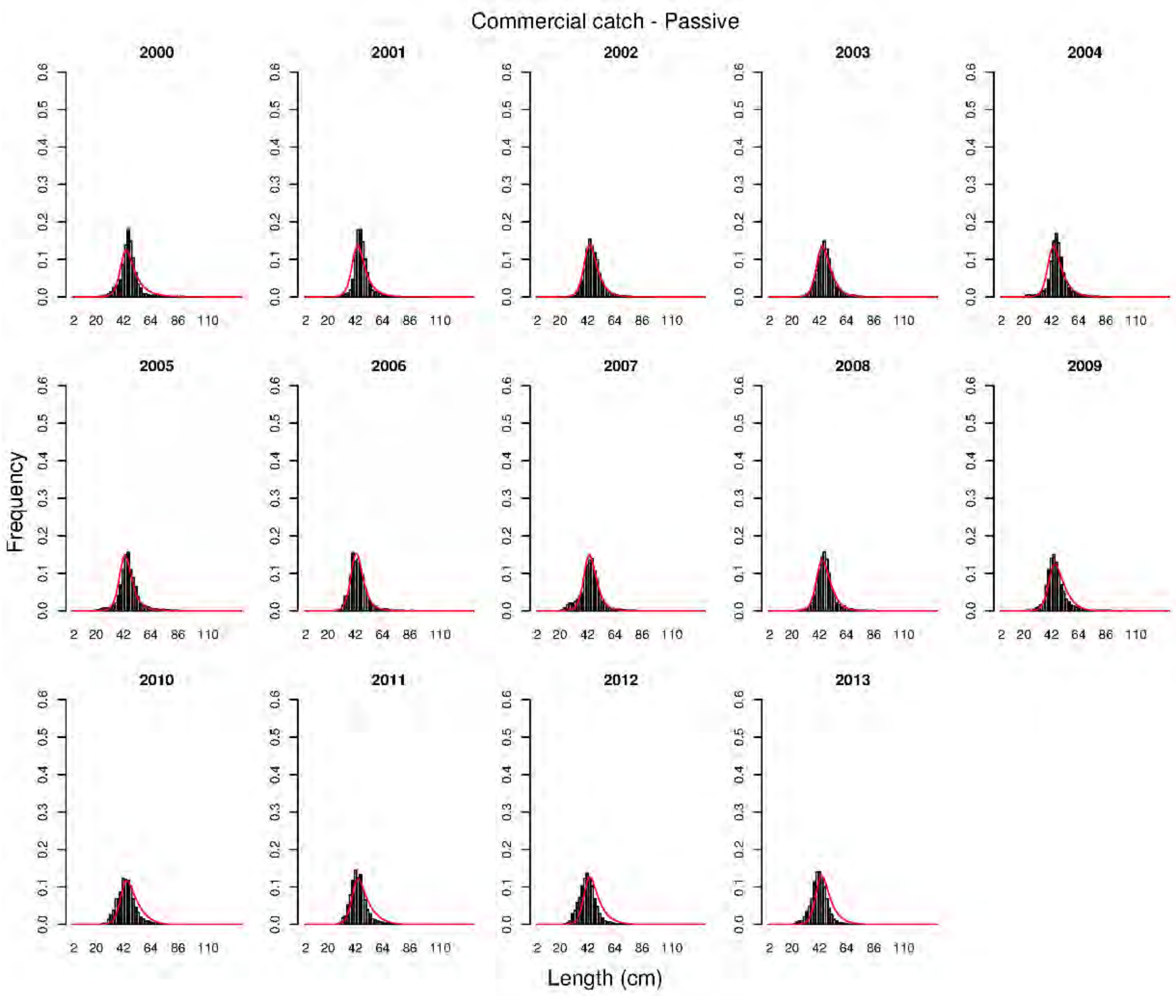

Figure 1.19. Observed (bars) and predicted (red line) annual average length distribution of cod from the commercial catches of passive gears in 2000-2013.

The length distributions of the two BITS surveys appear more problematic given the erratic appearance of a secondary peak. The modelled length distribution of the survey is centred around the main peak at approx. 25-30 cm in most of the years. The secondary peak of small cod around 10 cm has an inconsistent occurrence in the observations likely as a result of low selectivity, timing of the survey in relation to recruitment and variability in the growth of juvenile fish. As a consequence this secondary peak is not well represented by the model. The modelled survey indices of abundance capture the overall patterns observed by the BITS but the high variability in the index results in high residuals during certain years, ie underestimation of the abundance for most length groups around 1980. The recent increase in abundance is independently represented by the BITS surveys in both quarter 1 and 4 but the extent of the increase remain uncertain from the data. For this reason the increase of 20-40 cm cod appears under-estimated for BITS1. Age-length distributions are well represented by the model especially throughout the 1990s and 2000s. Growth appears somehow over-estimated in the last few years of the time series (2012-2013) due to a strong rapid reduction in the observed growth. This may contribute to explain the overestimation of the catches of the active fleet for the period 2004-2013 which severely affects the current model. In practice, to compensate for a too high weight of individual fish and still provide a good fitting of the survey indices the model would be forced to overestimate the removal of biomass by the fishery. We are currently working on the issue by developing a new likelihood function for weight-at-length data which was missing in the available version of Gadget. Results from new runs were not consolidated at the time of this report and are not included here. The estimated spawning stock biomass (SSB) is in good agreement with estimates from the last accepted assessment of the eastern Baltic cod stock. Also recruitment and fishing mortality show an overall agreement with the assessment but their interannual variability is estimated to be higher in Gadget most likely as the result of different assumptions between the two models (ie, the assessment model SAM assumes a random walk on F).

Multispecies model

The predator-prey interaction represented in our cod-herring-sprat model is regulated by two main aspects, the consumption and the prey size selection. Both processes are highly relevant in size structured multispecies models but also very uncertain. Different consumption rates have been proposed for cod using data from different areas and applying empirical models based on different assumptions. Bogstad and Mehl (1990) investigated the impact of alternative gastric evacuation models for cod in a multispecies framework (MULTSPEC) and showed how different models and assumptions could result in largely different estimates of prey consumed.

Different methods exist for the estimation of energy intake required for different activities in the life of fish. Data on stomach weights have been used to estimate how much food is eaten by accounting for the rate of evacuation of food from the stomach (see D2.2 and D5.2 for more details on the cod stomach data).

Attempts to estimate the same set of parameters of the single species models into the multispecies model have not been satisfactory so far. For this reason, the current multispecies implementation is largely based on same settings and parameters estimated during single species runs. Only exception is the natural mortalities (M) of the clupeids which have been estimated to be downscaled to account for contribution from cod predation. For herring the entire vector of M is reduced of 20% with the exception of age0, while the vector of M for sprat is simplified by a constant M corresponding to a reduction of 12-18% from the vector used in the single species model. Selected groups of parameters have been estimated in different runs of the multispecies model with the only purpose to verify that single species estimates still hold in the multispecies implementation.

Gastric evacuation model and maximum consumption

The gastric evatuation model proposed by Jones (1978) and derived from empirical studies on cod in the North Sea was used. To estimate the daily evacuation rate (R) the modelled was applied to the cod stomach data from the Baltic:

$$R = 0.16 * (L / 40)^{1.4} * S^{0.46} * 24 \qquad (11)$$

where:

< L > is the cod length < S > is the stomach weight

The model for the maximum consumption (M) implemented in Gadget is

$$M(L) = m0 * dt * e(m1 * T - m2 * T^3) * L^{m3} \qquad (12)$$

which was simplified assuming no effect of temperature (m1=m2=0). The parameters m0 and m3 were estimated using the consumption rates calculated from the Jones's gastric evacuation model. The parameters m0 and m3 were calculated based on quantile regression. Although the maximum consumption regulates the maximum amount of prey that cod is able to eat in a certain time interval (expressed in KJoules) we decided to select the 0.5 quantile rather than a higher quantile as the estimates appear more in line with daily consumption estimates from a number of experimental and field work (Jobling 1988; Uzars, pers. Comm.). Very little is known about other highly correlated parameters of the consumption such as the "half-feeding" value which was fixed to 0 and the otherfood component which was fixed to 1e10. The issue will certainly require further work at both the level of data and on the implementation of consumption in Gadget. The estimation of the maximum consumption parameters, assuming no effect of temperature resulted in m0 = 2.4e-02 and m3 = 2.96.

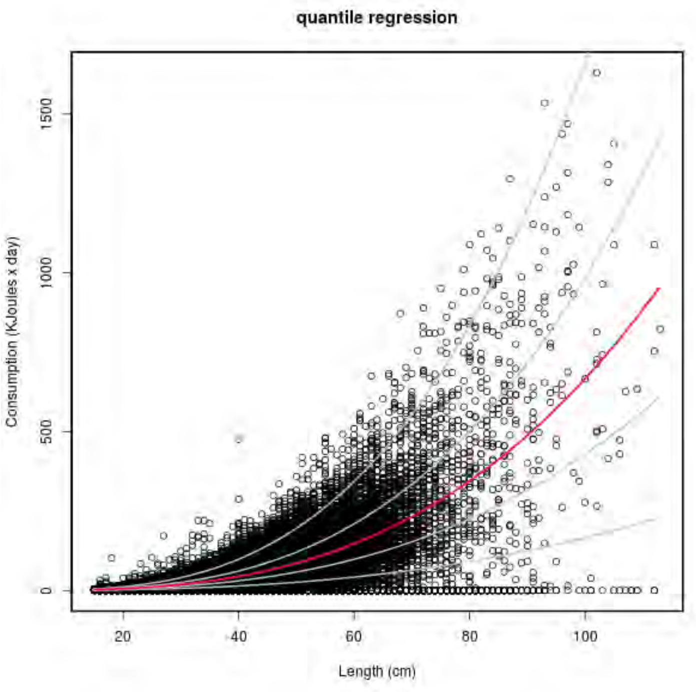

Figure 1.20. Cod consumption in relation to cod size as estimated from the Jones' gastric evacuation model. The grey lines represent the 0.05, 0.25, 0.75, 0.95 quantiles and the red line the 0.5 quantile used to implement cod consumption.

Predator-prey size selection

A marked relationships between the size of predator fish and the size of their preys have been documented for many fish species. During their ontogeny, most fish species not only are able to prey on progressively larger preys but also show an increasing prey size range. In practice, rather than simply shifting selectivity towards larger preys, many predators have an increasing size spectrum of preys. This pattern has been reported also among gadoid fish but it is only marginally observed in the cod stomach data from the Baltic.

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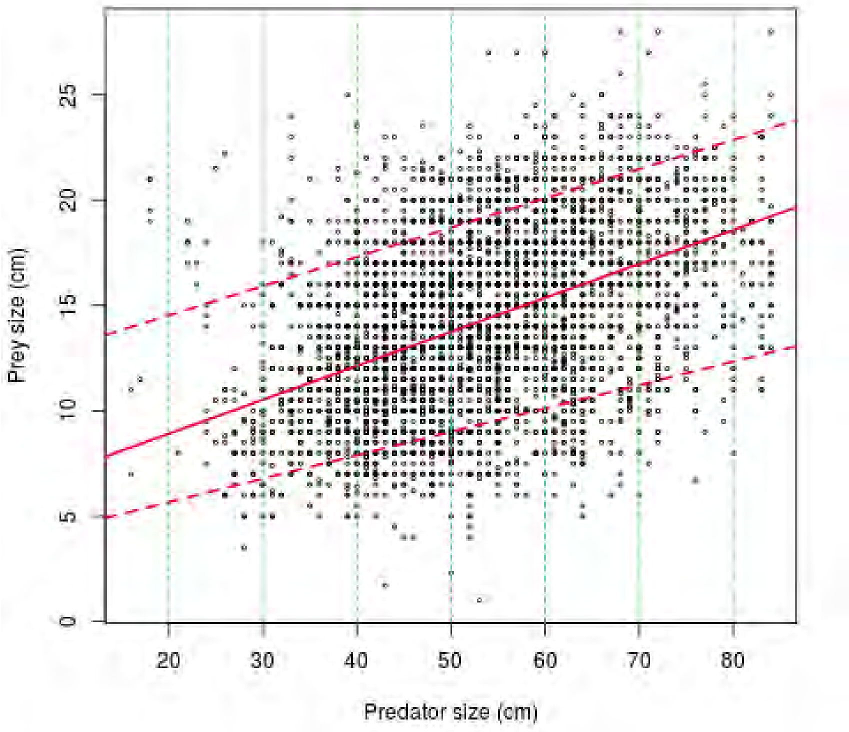

Figure 1.21. Scatterplot with cod and prey size with overimposed 10% and 90% quantile regression lines.

The selection applied by the predator towards for different prey sizes is prey specific and represented in Gadget by the Andersen suitability function. This function is characterised by the ratio of the predator/prey lengths which allows for changes in the prey size selection as the predator size increases. A simplified version of the Andersen function was adopted:

$$S(l,L) = p0 + p2\, e(-(\ln(L/l) - p1)^2 / p4) \text{ if } \ln(L/l) <= p1 \qquad (13)$$

$$S(l,L) = p0 + p2\, e(-(\ln(L/l) - p1)^2 / p3) \text{ if } \ln(L/l) > p1$$

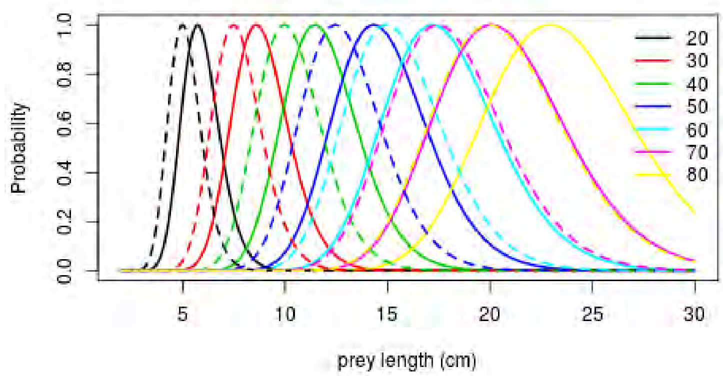

Figure 1.22. Predator-prey (cod-clupeids) size selection. Different colors refer to cod of different size. Continuous lines based on parameters from Trenkel et al. (2004) and dotted lines as estimated here.

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For exploratory purposes and as a guidance for the first selection of the suitability parameters, 3 of the parameters were estimated outside Gadget for a simplified version of the Andersen function (p0=0; p2=1) by assigning approximate selection probabilities to the 10% (0.1), 25% (0.5), 50% (1.0), 75% (0.5) and 90% (0.1) quantile regression. Estimated values of p1=1.390, p3=0.050, p4=0.049 were very close to the values used by Trenkel et al. (2004) for cod in the Celtic Sea (p1=1.25, p3=0.05, p4=0.05).

Other consumption parameters used in Gadget

To specify how much sprat and herring is consumed by cod in Gadget several other parameters are required the determine the other equations of consumption:

- the "half feeding" parameter (H) which specifies the biomass of prey required by the predator to consume prey at half of its maximum consumption level.

$$phi(L) = sum_p\ (Fp(l,L)) / (H\ dt + sum_p\ (Fp(l,L))) \qquad (14)$$

where:

< F > is the energy amount associated to the available preys suitable for cod

< dt > is the length of the timestep

By setting the parameter H to 0 we fix the feeding intensity of the predator which becomes independent from the preys abundance

the preference parameter of the predator for the prey (d) which controls the form of the functional response has been set to 1 for all preys which according to the user guide (Begley 2012) should correspond to a type II functional response.

$$F_p(l,L) = (S(l)\ E_p\ N(l)\ W(l))^{dp} \qquad (15)$$

where:

< S > is the suitability function

< E > is the energy content of the prey

< N > is the number of prey in the length cell

< W > is the mean weight of prey in the length cell

< d > is the preference of cod for each prey

Implementation of consumption in Gadget

Ultimately, the consumption of preys in our model is dependent on the size of the predator, and size and abundance of preys. For simplicity, the specific configuration of our model (see above) does not allow for the effect of temperature on the consumption and for changes in feeding intensity as a result of prey abundance (ie, prey density effect on consumption). Eq. 13-15 come into the consumption model implemented in Gadget:

$C_p(l,L) = N(L)\ M(L)\ phi(L)\ F_p(l,L) / sum_p\ (F_p(l,L))$ (16)

where:

< N > is the number of cod in the length cell

Likelihood components

Cod diet composition is informed in the model by the cod stomach data. The ratio of the presence of herring and sprat in the stomachs is calculated for four cod length groups 20-40 cm, 40-60 cm, 60-80 cm, 80-100 cm by 1 cm length aggregation of the preys. In practice, the model compares the observed and modelled ratio of stomachs with sprat and herring of different length in different size classes of cod. The likelihood function is represented by a simple sum of squared errors as follows:

$l = sum_t\ sum_p\ (P_{tp} - pi_{tp})^2$ (17)

where:

< P > is the observed ratio of the stomachs in each year and time step with a certain prey/predator length combination

< pi > is the ratio of the modelled consumption in each year and time step with a certain prey/predator length combination

Fitting

The modeled indices of abundance for the clupeids (BIAS) and cod (BITS) maintain a similar general good agreement with the observations.

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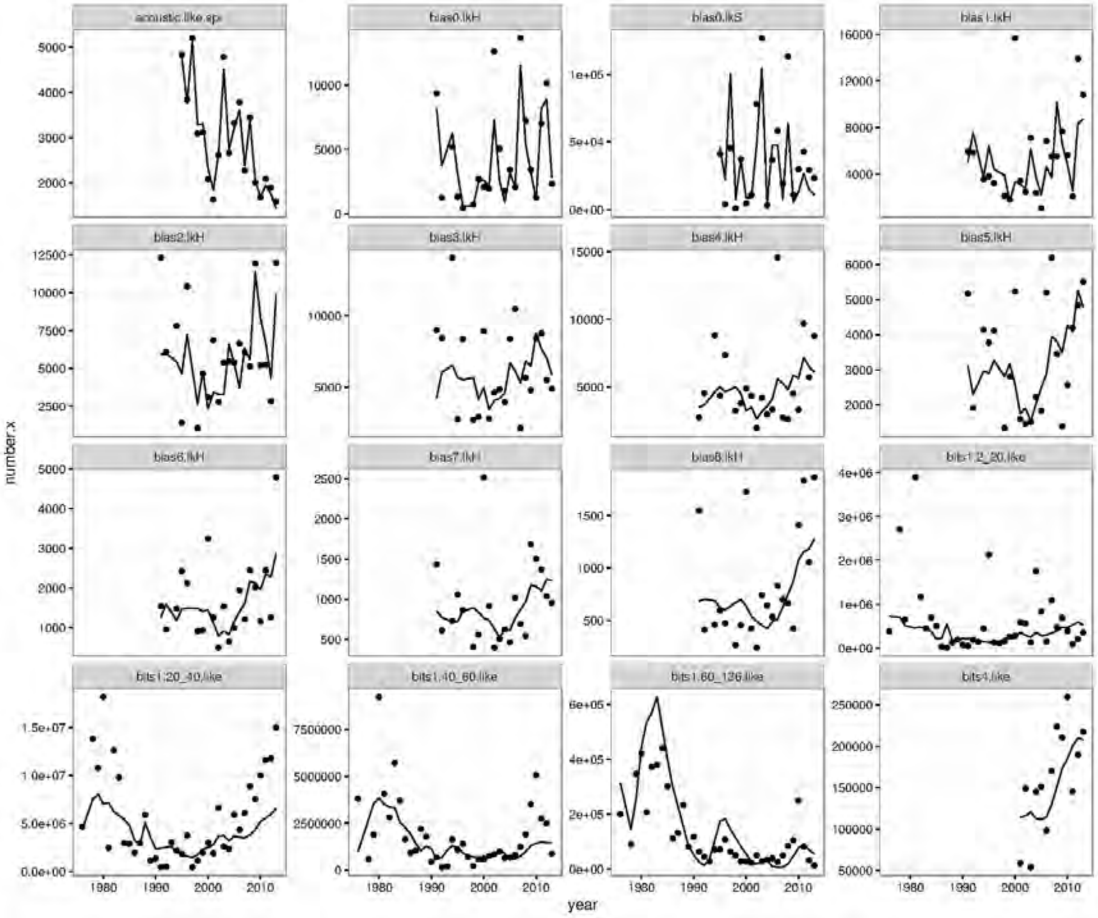

Figure 1.23. Observed (dots) and predicted (lines) indices of abundance for the clupeids (BIAS) and cod (BITS.

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Figure 1.24. Observed proportion of herring and sprat in the cod stomachs (dots) and consumed in the Gadget model (lines) for different cod length groups (20-40 cm, 40-60 cm, 60-80 cm, 80-100 cm).

Stock dynamics

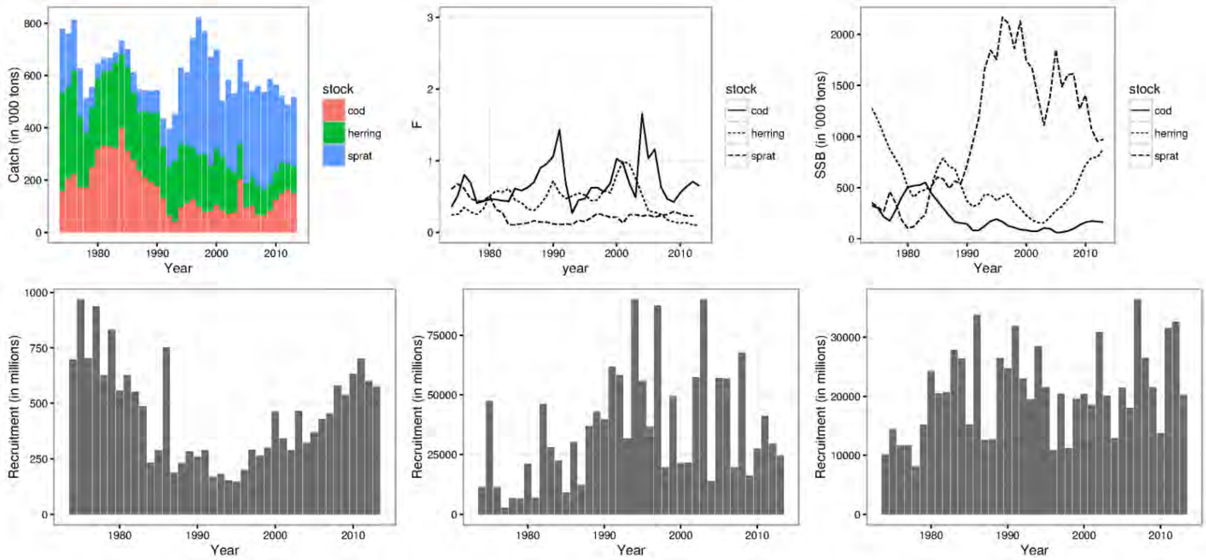

Figure 1.25. In the top row from left: total catches, fishing mortality and spawning stock biomass by species. In the bottom row from left: recruitment estimates of cod, sprat and herring.

Multispecies stock production model (MSPM)

Three species dominate Baltic fishery: cod, herring, and sprat. Species interactions occur mainly through cod predation on herring and sprat, and to some extent on young cod (cannibalism). In standard ICES stock assessment the effect of cod predation is included but not directly. First (once for three years), multispecies age-structured model (SMS, Levy and Vinther, 2004) is applied to provide estimates of herring and sprat predation mortality from cod. Next, these estimates are used in single species assessments of herring and sprat. Assessment of cod stock assumes constant natural mortality, although cannibalism in cod is well documented and quantified.

Growth of cod, herring and sprat underwent huge changes in recent decades; as result weight at age of these species in recent years has been about 40 – 60% lower than in 1980s or beginning of 1990s (Fig. 1.26). Neither growth changes nor predation mortality are modelled in standard ICES stock predictions; usually in predictions weight at age and predation mortality are assumed as average of recent values used in stock assessment.

For years it were problems with the assessment of the eastern Baltic cod stock, mainly due to difficulties and inconsistencies in age determination; these difficulties have been not resolved even if special study groups to deal with the issue have been set. The problems with Baltic cod assessment increased in recent years following deterioration of some input data (including further deterioration of age reading) and changes in environmental and ecological conditions for the stock (Eero et al., 2015). Consequently, since 2014 ICES has not been able to provide analytical assessment of the stock and advice on fishing opportunities basing on such assessment. The stock advice is based on survey trends only.

In such a situation the multispecies stock-production model (**MSPM**), which is using age information only partly, may overcome to some extent effects of inconsistencies in age determination on assessment performance. In addition, the model takes directly into account predation of cod on herring and sprat which is important element of species interactions in the Baltic. Cannibalism in cod is also simulated in the MSPM what is not the case in the ICES assessment.

Thus, the multispecies stock-production model (Horbowy 1996, Horbowy 2005) was further developed and tested as a tool to simulate stock dynamics and multispecies interaction in the Baltic. In classical production models fish growth is assumed constant. In the Baltic, growth of three main species (cod, herring, and sprat) has shown marked changes/declines within three recent decades and it is necessary to model such changes to adequately reproduce stocks dynamics. Therefore, the sub-models of cod herring, and sprat growth were developed and included into new version of the multispecies stock-production model. The growth was modelled as:

- related to area of hypoxic waters in case of cod,
- dependent on salinity for herring,
- stock-density dependent for sprat.

In addition, new stomach contents data covering 1994-2014 and compiled within MareFrame were included into the model and with sub-models of herring and cod, herring, and sprat growth form new basis of the model. The model was fitted to analytical estimates of recruitment and fishing effort (taken as estimates of recruitment and fishing mortality from ICES analytical assessments) to inspect model behaviour in case when good indices of effort and recruitment are available.

Several runs of the MSPM on historical part (1982-2013) were performed to parameterise the model. The model parameters were estimated by minimising the sum of squared differences between modelled and observed catches and modelled and observed stomach contents of cod; the logarithmic scale was applied to calculate the differences.

The parametrised model was the basis for the development of the prediction model in which scenarios of Baltic fish stocks development under different environmental conditions and fishing intensity can be simulated. The prediction part of the model includes dependence of cod growth on area of hypoxic waters, dependence of herring growth on salinity, and dependence of sprat growth on stock density. Stochasticity may be included in the predictions by disturbing initial biomasses and stock - recruitment relationships with log-normal random errors (measurement and process error, respectively). In the present implementation of the model the prediction may be run till 2100 with the defined by the user range of fishing mortalities.

Methods and data

The multispecies stock-production model

In the model the change of biomass B is

$$\frac{dB_s}{dt} = (v_s h_s w_s^{-1/3} - k_s - q_s E_s - M1_s - \sum_{r=1}^{n} h_r w_r^{-1/3} \frac{G_r^s B_r}{\sum_{j=1}^{n} G_r^j B_j + OT}) B_s \qquad (1)$$

where

B_s = biomass of stock s,

v, h, k = parameters of the von Bertalanffy growth equation generalised by Andersen and Ursin (1977), v is the fraction of eaten food assimilated for growth,

E = fishing effort,

q = catchability coefficient,

M1 = coefficient of natural mortality caused by reasons other than predation (residual natural mortality) ,

w = mean weight of fish in the stock,

G_r^s = suitability of prey s to predator r,

OT = "other food"

s, r, j = stocks,

n = number of stocks.

The last expression in the brackets (starting with first summation symbol) represents predation mortality (M2). Assuming that the term in brackets is constant or has low variability in a time interval [t, t + dt], model (1) can be approximated by

$$B_s(t+dt) = B_s(t)\exp[a_s(t)dt], \tag{2}$$

where

$$a_s(t) = v_s h_s w_s^{-1/3} - k_s - q_s E_s - M1_s - \sum_{r=1}^{n} h_r w_r^{-1/3} \frac{G_r^s B_r}{\sum_{j=1}^{n} G_r^j B_j + OT} \tag{2a}$$

If recruitment takes place at time t+dt, equation (2) will take the following form

$$B_s(t+dt) = B_s(t)\exp[a_s(t)dt] + R_s \tag{3}$$

where R is the biomass of the year-class recruited to the stock.

Additional equations are incorporated into the model to mimic unexploited/young ages in the modeled stocks. If unexploited part consists of a few age-groups, it can be modeled using eq. (3) with fishing effort set to zero. However, in Baltic unexploited (low exploited) parts consist only of 1-2 age-groups. The dynamic of the unexploited part of the stock (assumed as age 0 and 1 for herring and sprat, and age 1 and 2 for cod) is presented by

$$N_{si}(t+dt) = N_{si}(t)\exp[-(M1_{si} + \sum_{r=1}^{n} h_r w^{-1/3} \frac{G_r^s B_r}{\sum_{j=1}^{n} G_r^j B_j + OT})dt] \tag{4}$$

$$B_{si}(t) = N_{si}(t) w_{si}(t)$$

where

N_{si} – number at age i in stock s,

w_{si} – mean weight at age i in stock s.

The part of the oldest unexploited (young) age which survives until being exploited leaves the unexploited component and enters the exploited component of the stock as recruitment denoted by R_s in equation (3).

Recruitment to the unexploited component of the stock may be modeled using classical stock-recruitment (S-R) functions or may be implemented as

$$R_0 = uR_{index}$$

where u is some parameter, and R_{index} – is an index of recruitment (e.g. from survey) to the unexploited component.

Parameterization of the model and application to the central Baltic (Subdivision 25-32)

The model was applied to simulate the stock dynamics and trophic interactions of cod, herring, and sprat stocks in the central and eastern Baltic (Sub-divisions 25-32) in 1982-2013. Adult stock of cod was comprised of fish at age 3 and older, adult stock of herring and sprat were assumed to be at age 2 and older. Cod ages 1-2, and herring and sprat ages 0-1 were included in the model as young fish components. Simulated trophic levels are presented in Fig. 1.26. Adult cod eats everything in the system, young cod eats young herring and sprat and adult sprat. The fishery is on top of the system exploiting adult cod, herring, and sprat.

Following described changes in cod, herring, and sprat growth described in the Introduction, the growth was related to environmental variables (cod and herring) or made density-dependent (sprat). The following exploratory variables were used: area of hypoxic waters for cod, salinity for herring, and stock-density for sprat. These dependencies are quite strong and above variables explain from about 60 to almost 70% of the growth variance (Fig. x3 a, b, c). In the MSPM growth of fish (anabolism rate) is represented by parameter h (eq. 2a) and this parameter was made dependent on above variables after relevant transformation from weight at age terms to the h.

To run the model fishing effort and recruitment indices or sub-models for recruitment are needed.. Fishing mortality estimates from ICES assessment (ICES, 2013) were taken as fishing effort indices, and similarly the XSA estimates of the youngest ages numbers were used as recruitment indices. Such approach represents situation when relatively good quality data on fishing effort and recruitment are available.

The unknown parameters of the model were estimated by minimization of the sum of squared differences between observed and modeled values, i.e.

$$SS(G,q,u,B0) = \sum_{s,t} \lambda_Y (\ln Y_{st} - \ln \underline{Y}_{st})^2 + \sum_{s,t} \lambda_{sc} (\ln SC_{st} - \ln \underline{SC}_{st})^2 + \lambda_{\mathbf{B}} (\ln B0_s - \ln \underline{B0}_s)^2 \quad (4)$$

where Y and $\underline{Y}$, SC and $\underline{SC}$, and B0 and $\underline{B0}$ denote observed and model catches, stomach contents, and initial biomass, respectively. Index s refers to species (cod, herring, sprat) and t is year (1982-2013). The parameters λ represent statistical weights which were the inverse of the variance associated with successive residual terms. The parameters G are determined relative to a constant multiplier, so the highest was allotted 1, and other G values were estimated relative to that. The other food (OT) was assumed constant at 1000 units.

The model was validated through inspecting the distribution of residuals and retrospective analysis.

Predictions with the model

The estimates of adult stock size and recruitment for the beginning of 2011 were used as staring values for deterministic predictions with the model. In historical part of the model adult stock size was calculated for 2013 but last recruitment estimates were available for 2011, so that year was a starting year for predictions. Predictions were performed till 2100, and growth rate was assumed as average of recent 5 years values. In predictions for herring and sprat the Beverton and Holt stock-recruitment relationship was assumed, and its parameters were estimated by fitting that relationship to estimates obtained from the multispecies production model for 1982-2011. In case of cod the Ricker S-R function was used. The S-R data for cod were constrained to period from 1987 onwards, to take into account regime shift in the Baltic and changed productivity of the cod stock.

In the deterministic predictions fishing mortality for all species ranged from 0 to 1.4 with step of 0.1. Predictions comprised all combinations of such fishing mortalities for cod with Fs from similar range for clupeids (to reduce total number of simulations Fs of herring and sprat were assumed the same in a given simulation), giving in total 15*15=225 combinations of cod and clupeids fishing mortality. That allowed determining range of fishing mortalities associated with MSY for cod and clupeids.

Results

The stock dynamics and estimates of multispecies interactions

The model fits quite well to catch data and standard deviation of the residuals of logged catches (or CV of estimated catches) varies from 0.17 to 0.23. However, the fit to stomach contents data is worse and the variance of estimated food consumption is much higher (CV mostly varies from 0.6 to 0.9).

Biomass estimates from MSPM are quite similar to biomass estimates from standard ICES assessment and SMS assessment (Fig. 1.29). The differences between average biomass from MSPM and ICES assessment are 6% for cod and sprat and 10% for herring. Somewhat higher disagreement for herring is result of major differences in biomass estimates in first half of time series; in recent years the MSPM and ICES estimates of herring biomass are very similar.

Predation mortality of all considered species declined strongly in 1982-1992 following decline of cod stock. Next, M2 have fluctuated at rather low levels and showed some increase at the end of last decade. That increase was associated with some increase in size of cod stock. When fish preys are considered, the highest was M2 of young herring and the lowest of adult herring. Predation mortalities of young and adult sprat were similar and between herring values of M2. (Fig. 1.30).

Biomasses of herring and sprat consumed by cod show similar dynamics as predation mortality of prey species. Very high consumption of sprat in 1995 was effect of both increase in cod biomass and very strong sprat year class of 1994. On average cod consumed about 65% more sprat than herring and total consumption of fish was very similar to consumption of other food (Fig. 1.30).

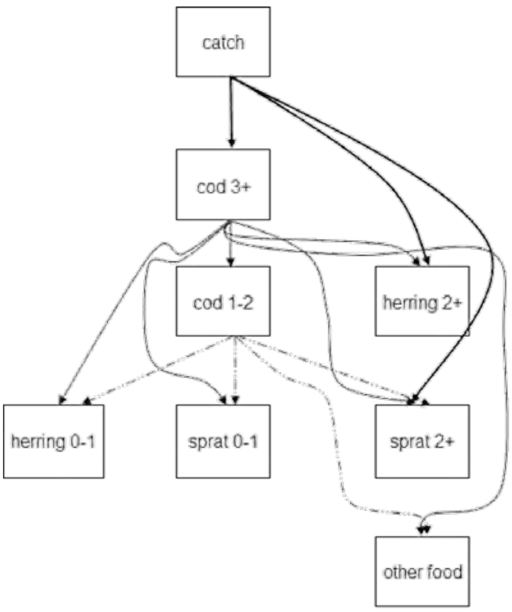

Fig. 1.26. The trophic interactions in the Baltic simulated in the multispecies stock-production model.

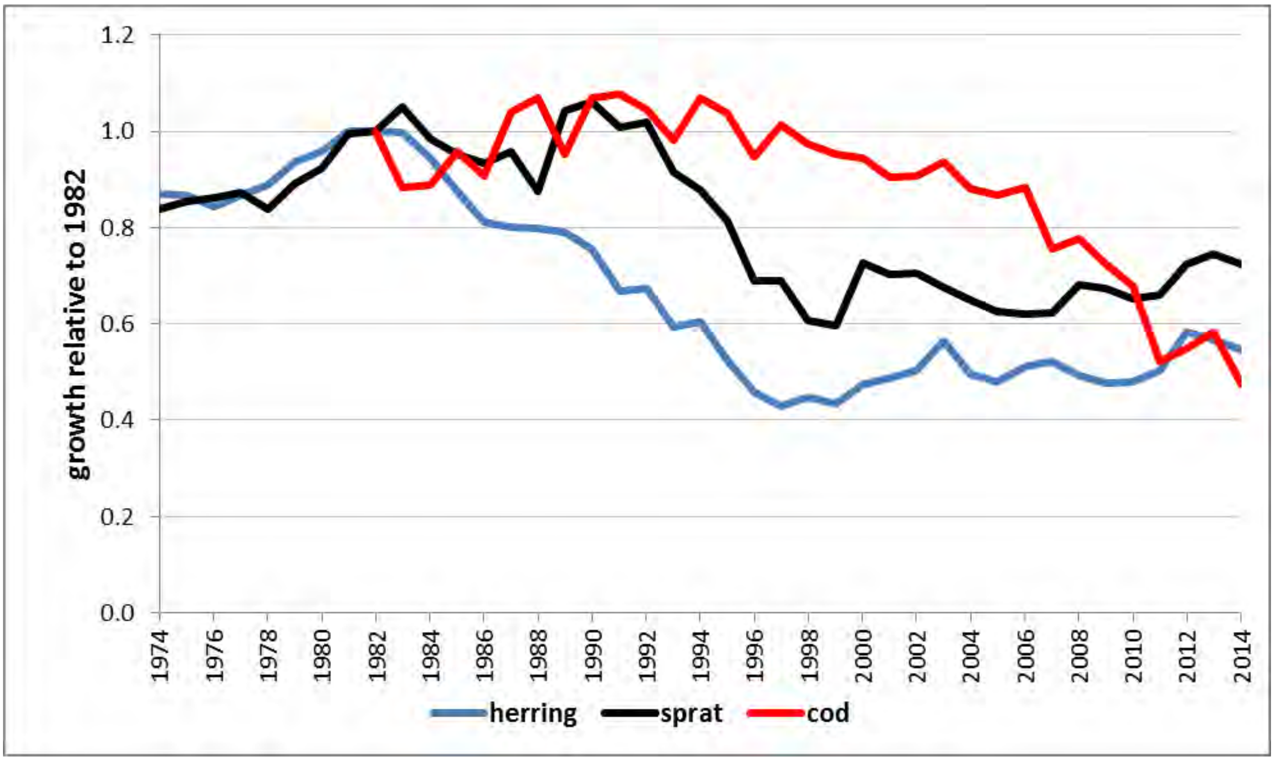

Fig. 1.27. Growth of cod, herring, and sprat in relative terms (1982=1)

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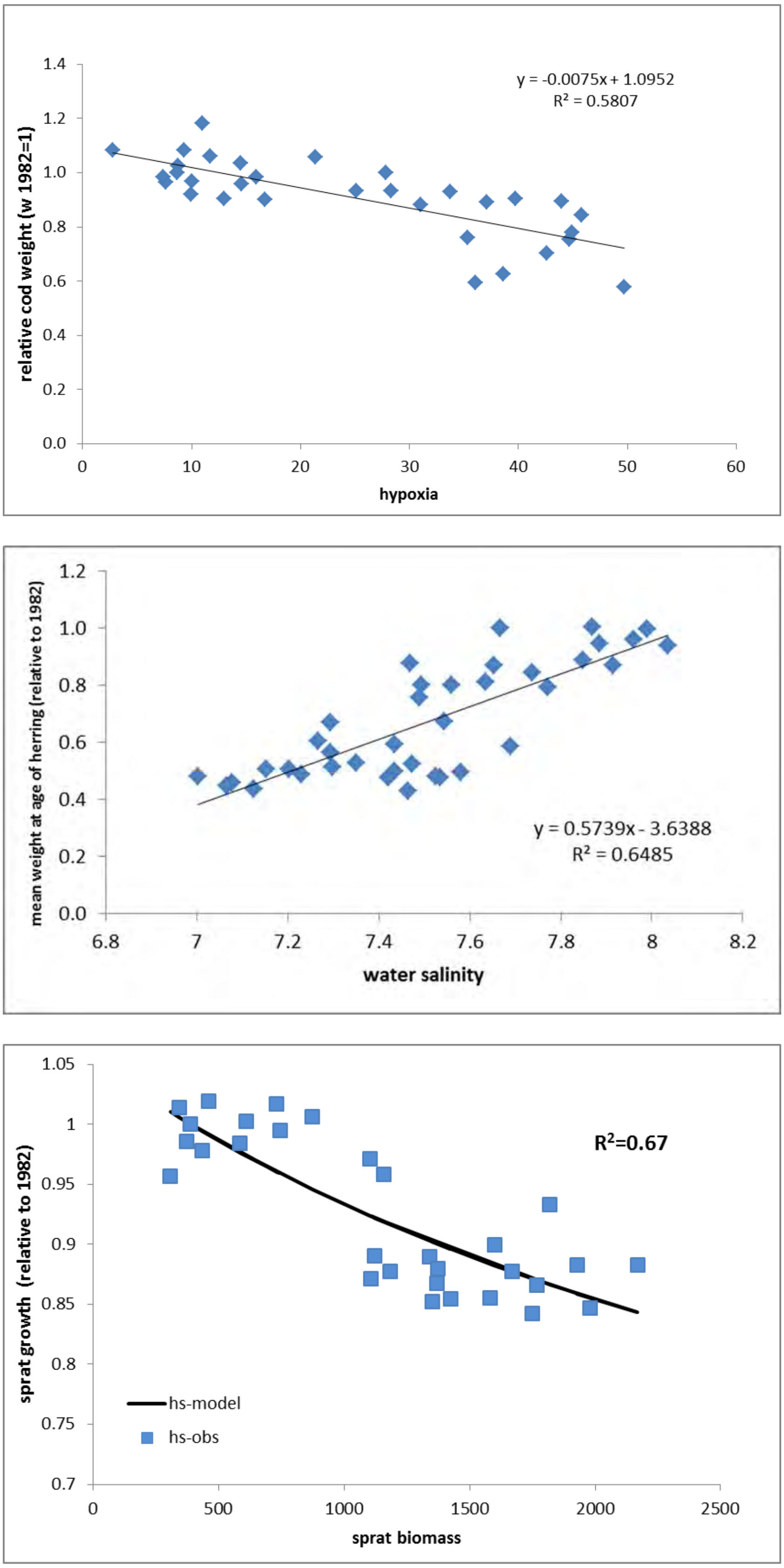

Fig. 1.28. Dependence of cod, herring, and sprat growth on area of hypoxic waters (cod), water salinity (herring), and stock density (sprat). Linear models fitted to cod and herring growth, hyperbolic model for sprat density dependence.

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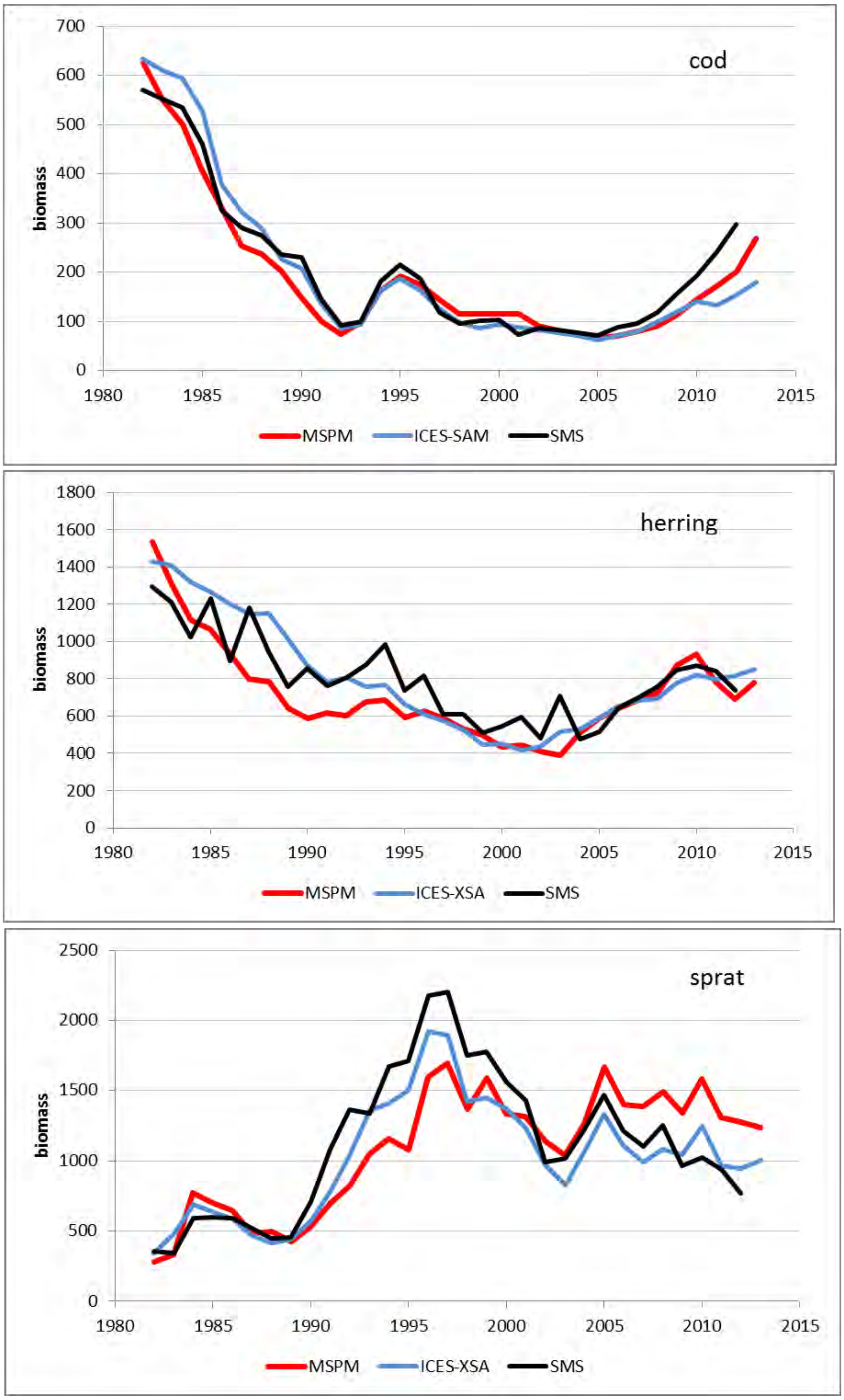

Fig. 1.29. Comparison of estimates of biomass from the MSPM, ICES assessment (SAM or XSA) and ICES assessment with SMS.

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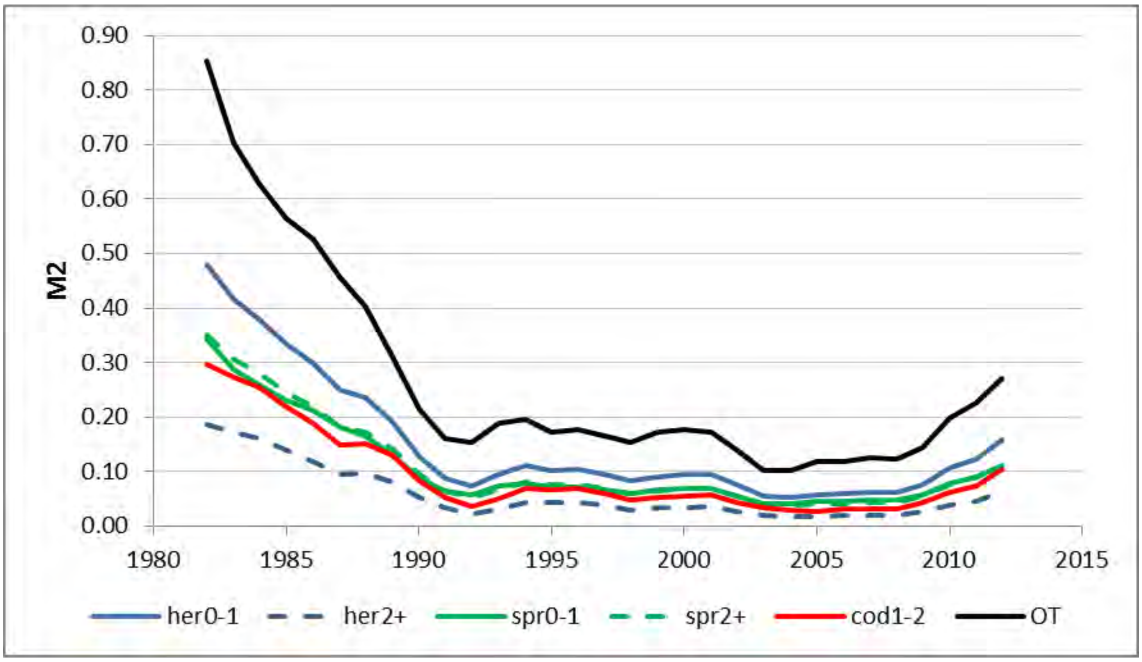

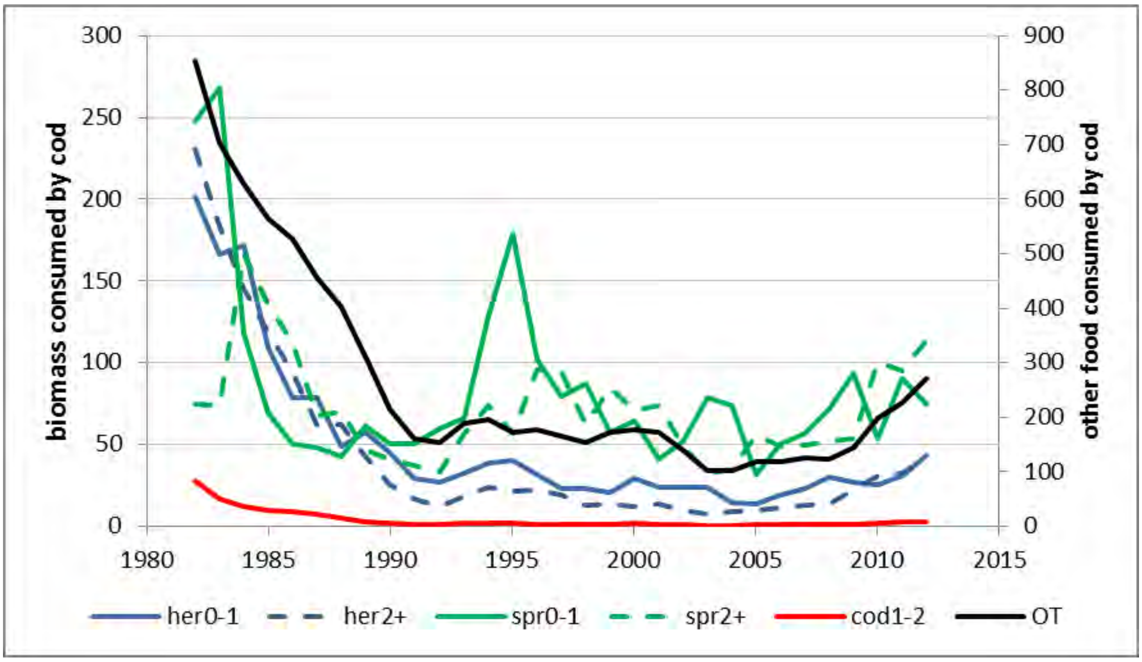

Fig. 1.30. Predation mortality and biomass consumed by cod. Prey species separated into young and adult components (e.g. spr0-1 and spr2+, respectively). OT is other food. Predation mortality has meaning of M2 by ages weighted by prey biomass.

Scenarios evaluation

The Baltic Sea case study aims to evaluate the performances of a set of alternative management options which have been designed considering feedback from the stakeholder group, limits imposed by the models adopted, and in relation to a set of objectives and criteria in part derived from the current ICES approach, the CFP and the MSFD. This work has produced six fisheries management scenarios for testing. These scenarios have been run in combination with different climate (2), nutrient load (2) and seal population growth (2) scenarios which altogether generate an envelope of uncertainty around the predicted ecosystem trajectories. The two climate scenarios include no warming andmoderate warming (A1B IPCC scenario). As climate models substantially differ in their forecasted warming trajectories, we simulated two 'versions' of the same (A1B) warming scenario, using outputs from two different climate models. The two nutrient management scenarios, which have an effect on the eutrophication of the basin and severity of oxygen depletion, contrast a regime of increasing nutrient loads with the expected regime under the Baltic Sea Action Plan. The two seal population growth scenarios investigated include 5% and 10% growth rates (Sara Königson pers. comm.). Fisheries management scenarios only differed from each other in the relative fishing mortalities applied on different stocks, no additional regulations were simulated. Fishing mortalities were kept constant after 2013 and were chosen to satisfy scenario criteria detailed below.

Business As Usual (BAU)

According to ICES recommendations for an FMSY approach for the eastern Baltic cod, the central Baltic herring and the Baltic sprat.

MaxYield Cod

Maximize total cod yield. Cumulative catches of cod until 2030 are maximized. At the moment this scenario is a replacement, and it is not a comparable substitute in any way to the scenario 'Cod MSY in a multispecies context' which was agreed upon during previous meetings with the stakeholders and foreseeably will be implemented for the second version of the Decision Support Tool. It has to be noted that this and all the following scenarios are not run to equilibrium, hence, sustainability in the strict sense of the word was not a criterion.

MaxProfit Cod

Maximize cumulative discounted profit of bottom trawlers and gillnetters based on their cod catches until 2030. Cod prices are based on Swedish sale notes (2011-2015) and are different for juvenile and adult cod but the same for the two fleet segments. Profits are also a function of seal abundance: increasing seal abundance decreases cod price, to represent the effects of seal damage. Costs are based on fishing mortality imposed on the stock and parameters are calculated separately for the bottom trawl (BT) and gillnet (GN) fisheries based on STECF data and fishing mortality assessments from the Gadget model. Discount rate is 3%.

MaxProfit Pelagic

Maximize cumulative discounted profit of pelagic trawlers (PT) based on their herring and sprat catches until 2030. Clupeid prices are based on Swedish sale notes (2011-2015) and are the same for juveniles and adults cod. Costs are based on amount landed and parameterised based on Voss et al. (2014) who used STECF data of pelagic trawler and seiner fleets (2002-2008).

MaxProfit Total

Maximize the sum of the latter two (BT + GN + PT)

MaxEnvironmentalState

Maximize cod biomass compared to clupeids (D/P ratio) but also keeping herring and sprat at viable levels (average SSB > Blim for sprat and average SSB >Blim for herring).

Ecopath with Ecosim (EwE)

All combinations of scenarios mentioned above (seal, climate, nutrient and management) have been simulated by EwE.

Model settings

In the case of the Business As Usual scenario ICES F_{MSY} values are used as fishing mortality forcing. These values are expressed as instantaneous fishing mortality, while in EwE fishing mortality is applied as annual yield/biomass. Thus, we rescaled ICES values using the relationship F_{EwE} = 1-e^(-F_{ICES}). We also needed to rescale B_{lim} values, to account for the fact that EwE 'adult' groups are just a proxy for SSB. That rescaling was done by calculating the statistical relationship (linear regression) between EwE hindcasts of adult biomasses and assessment SSB outputs and rescaling ICES values using the obtained intercept and slope. ICES reference values and those applied in EwE are compared in Table 1.8.

Table 1.8 ICES reference points and corresponding values applied in EwE.

	EwE	ICES
Fishing mortality cod	0.37	0.46
Fishing mortality herring	0.2	0.22
Fishing mortality sprat	0.23	0.26
B_{lim} herring (t)	582000	430000
B_{lim} sprat (t)	397000	410000

EwE does have inbuilt routines for optimization on certain criteria. However, these routines are not completely customizable. Thus, for the greatest possible comparability with the other two modelling approaches, we conducted all maximizations by a 'brute force' approach, that can theoretically be implemented in all three modelling frameworks used in the case study. This means that we simulated the combinations of all seal, nutrient and climate scenarios in combination with a range of fishing mortality (F=yield/biomass) forcing settings. The latter were defined as all factorial combinations of Fs of adult groups ranging from 0 to 1 in increments of 0.1. This resulted in a thousand F scenarios plus an additional one representing the Business As Usual scenario. Fishing mortalities of juvenile groups were a constant fraction of adult fishing mortalities, and flounder fishing mortality was a constant fraction of cod fishing mortality, as flounder is to a large extent a bycatch in cod fisheries. In both of these case we used the same fractions as in 2012-2013.

All forcing files (including fishing mortality forcing functions and environmental forcing functions) were generated in R and exported as .csv files for import into EwE. Simulations based on these files were run automatically by EwE using the MultiSim tool. As MultiSim is not able to vary biomass time series forcing, which we needed to simulate different seal scenarios, we modelled seal scenarios by generating two EwE model files, one including the low seal growth trajectory, one the high seal growth trajectory, and ran all scenarios with MultiSim twice, using the two model files. The resulting raw EwE output files (from in total 1001x2x3x2=12012 simulation runs) were imported into R, where yields, profits, SSBs and the demersal to pelagic ratio were calculated for all individual model runs. Then for each management, climate, nutrient and seal scenario combination, we selected the F combination that maximised the target defined by the management scenario (yield, profit or Demersal/Pelagic) and, in the case of the MaxEnvironmentalState management scenario, additionally satisfied criteria on minimum levels of herring and sprat SSBs. This procedure resulted in the 'optimal' sets of Fs for each management scenario, conditional on the climate, nutrient and seal scenario settings. We filtered individual run outputs to contain only those ran with optimal F's for each scenario combination, and calculated all indicators from these, for each simulation year.

An overview of the process detailed above is shown on 1.31.

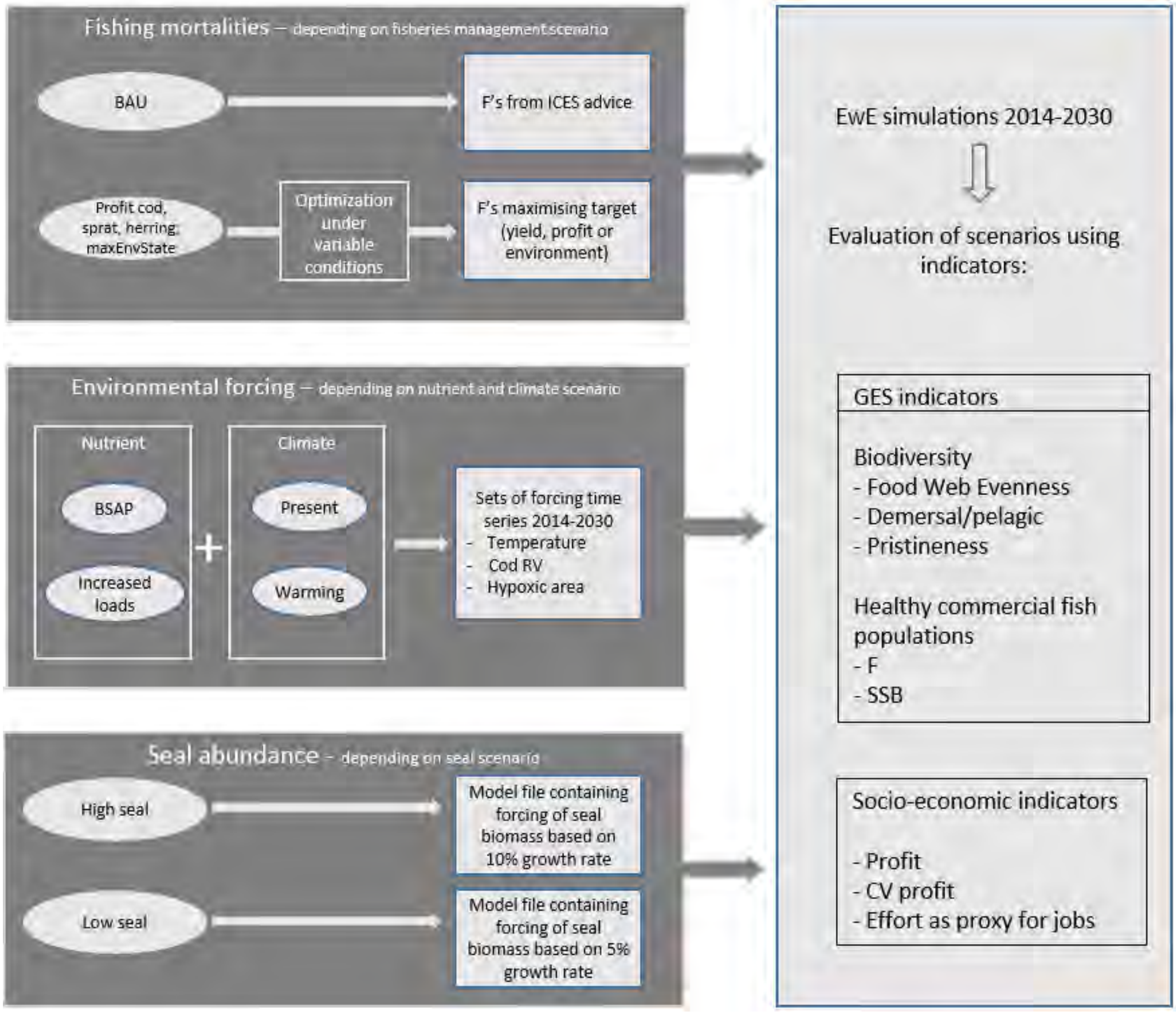

Figure 1.31. Overview of scenario simulations as conducted in the EwE modelling framework.

Scenarios output

Time series of the spawning stock biomass (SSB) of cod, herring, sprat and profits of the three modelled fleets (bottom trawls and gillnetters catching cod, pelagic trawls catching sprat and herring) in each management scenario are shown on Figure 1.32. Total stock biomass trends were very similar to those of SSB, therefore, they are not shown extra on any of the figures. Fishing mortality ranges applied are shown on Figure 1.33 and average indicator outcomes of different management scenarios on Figure 1.34.

The Business As Usual scenario, in which ICES recommendations are followed in setting fishing mortality levels, resulted in the least changes compared to current state. Cod and herring showed some decline, as well as the profits of the bottom trawl sector as fishing on cod is reduced. Herring fishing is increased which resulted in higher employment in the pelagic trawl sector.

The strategy maximizing cod yield was to almost cease fishing on sprat (a prey of cod which is though less preferred than herring, but more vulnerable to cod predation in the model) and fishing cod at a very high level. Herring fishing mortality depended on the environmental scenarios, the reason for which will be investigated further. It is probably related to the fact that relative strength of cod predation on cod versus competition between these stocks for benthic food depends on hypoxia and temperature levels. Thus, the effects of decreasing the herring stock may be beneficial or detrimental to cod yields depending on environmental conditions. High levels of fishing on cod naturally resulted in high levels of employment in the demersal fishing sectors, but they had detrimental effects on the profits in these sectors as well as on both on cod and herring biomasses. Sprat increased the most in this scenario among all scenarios.

A strategy maximizing cod profit instead of cod yields had the opposite effects on the demersal sector. In this case, cod and sprat fishing levels were kept low which allowed a moderate amount of increase in cod. For the same reasons mentioned before, herring fishing levels were variable. This strategy resulted in increasing profits for the demersal sector, but decreasing employment and some increase in cod as well as sprat.

Maximising the profit of pelagic trawls resulted in the largest decrease of total fish biomass, as in this strategy all stocks were fished at very high levels. A decrease in cod enabled all clupeid production to serve the fisheries, however, it had detrimental effects on profits in the demersal sectors, while employment was kept at current levels. Both profits and employment increased within the pelagic sector, as especially herring was fished to a larger extent than today.

Maximising total profit of the fishery differs from the previous strategy in that fishing on cod is lowered. This negatively affects the sprat stock, but less so the herring stock. Thus, this strategy still allowed for pelagic profits and employment to increase while profits of the passive demersal fleet also increased.

The 'Maximising Environmental Status' scenario aimed for increasing the ratio of cod compared to clupeids, without dangerously decreasing biomass of the latter. This was generally achieved by low overall fishing levels compared to other scenarios. However, the optimal fishing mortality on cod to achieve the scenario target strongly depended on the environmental scenarios considered. In some environmental scenarios, clupeids tended to strongly decrease, which meant that fishing mortality of cod needed to be increased to minimize its predation and help to keep these stocks from collapse. In other environmental scenarios the conditions for clupeids were better, thus, the fishing mortality of cod was lowered to reach the high Demersal/Pelagic ratio which was the actual target of the scenario.

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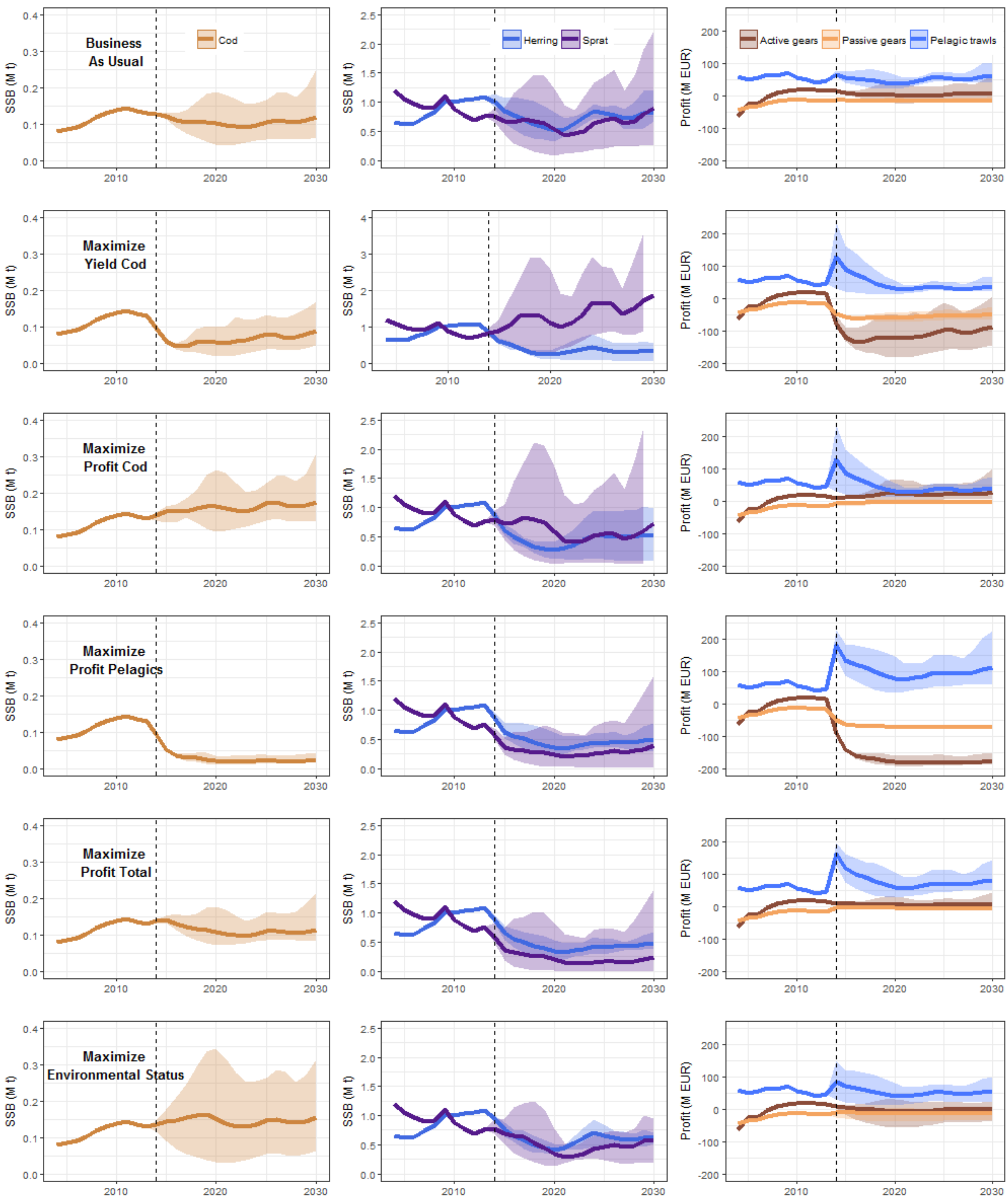

Figure 1.32. Time series of SSB of cod (left panel), sprat (purple) and herring (blue, middle panel) and profits of active demersal gears (brown), passive gears (orange) caatching cod and pelagic trawls (blue) catching herring and sprat. Vertical dashed line indicated start of forecasts. Solid lines are the mean forecast from all runs, ranges around the lines indicate maximum and minimum levels from all runs, thus, the level of environmental uncertainty. Each row represents a distinct management scenario, indicated in bold letters in the left panel. Please note the different y scale of pelagic fish biomass plot in the Maximize Cod Yield scenario.

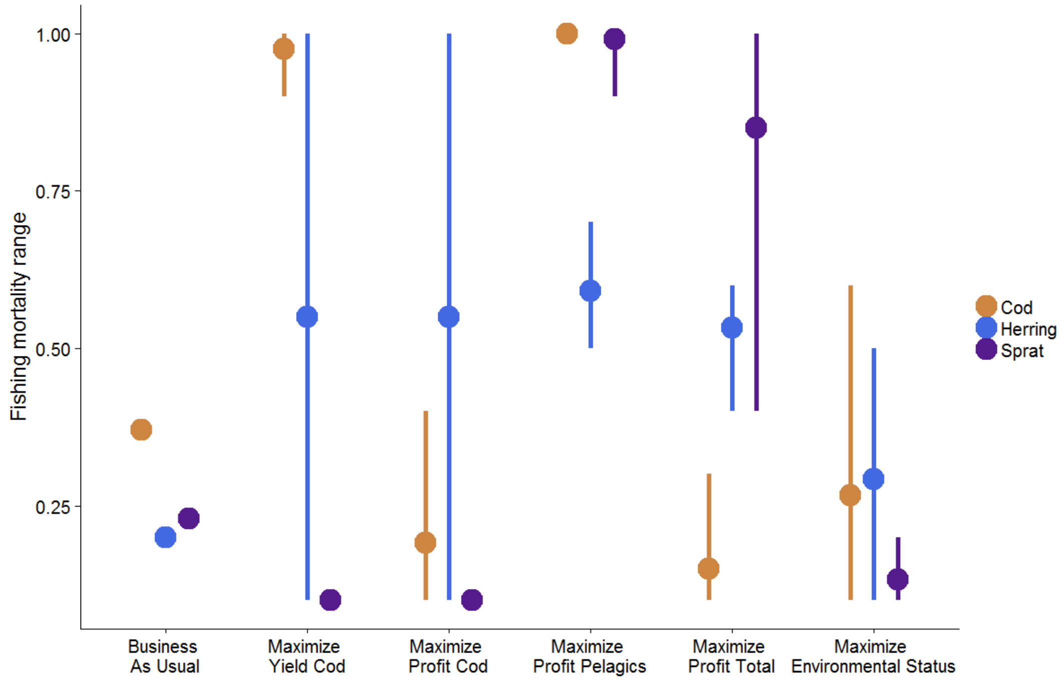

Figure 1.33. Fishing mortalities applied on cod (brown), herring (blue) and sprat (purple) in the six different management scenarios. Dots represent mean values, the extent of lines the total range, which represents the extent of environmental effects on what are optimal fishing levels to achieve the scenario target. There is no uncertainty in the Business As Usual scenario, as fishing mortalities were set and kept according to current ICES recommendations irregardless of environmental conditions.

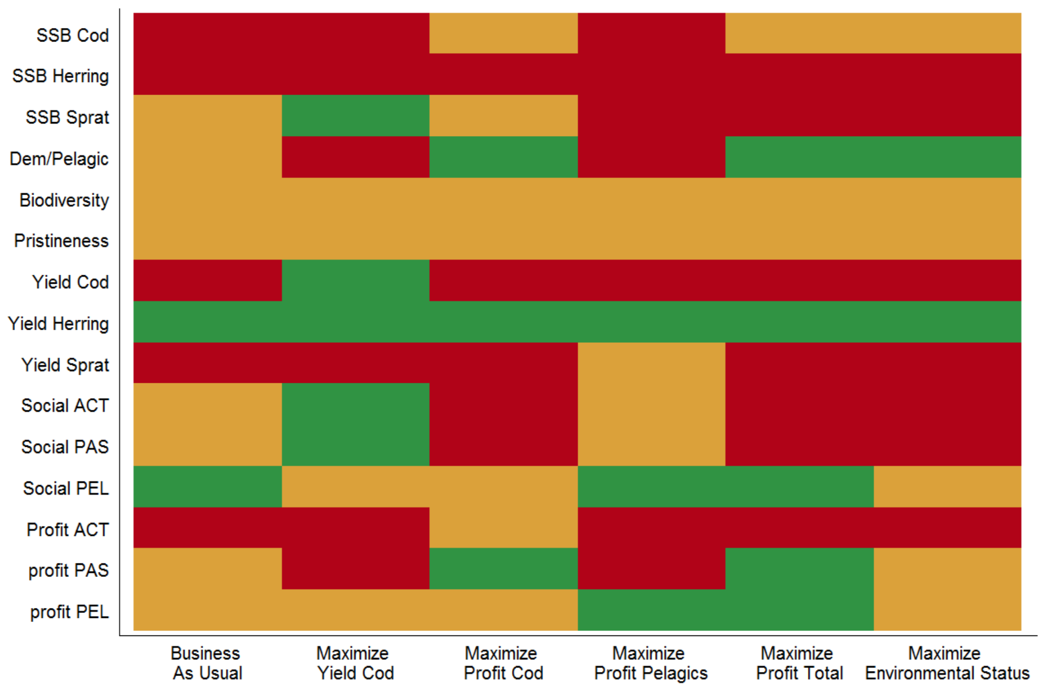

Figure 1.34. Average levels of selected indicators (rows) in six different management scenarios. Green indicates an average improvement compared to current conditions (more than 20% increase in mean indicator level compared to mean 2011-2013 level), yellow similar status to now (mean indicator level does not deviate from mean 2011-2013 level more than 20%), red a decrease (more than 20% decrease in mean indicator level compared to mean 2011-2013 level).

Gadget

At the current stage only the *BAU* scenario has been implemented in Gadget. Main limit to implementation of the other scenarios has been so far the lack of an explicit link in the model between cod growth and consumption of clupeids. The approach offered by Gadget which links fish growth to prey availability and feeding within the context of energy budget is attractive but it has proved to be extremely challenging in the implementation and examples of its use are extremely scarce in literature (Stefansson and Palsson, 1997). This is one of the main areas of model development on which we are currently investing time and resources but none of the attempts has resulted in satisfactory results so far. The feedback loop of the clupeids on cod growth and condition is of primary importance to capture the full implications of predator-prey interaction in the Baltic and its lack compromises a meaningful implementation of most of the other scenarios, with the possible exception of the *MaxProfit Pelagic* scenario. We also recognize that routines to project Gadget models into the future are still in their infant stage which has contributed to limit the implementation of the most complex scenarios, but some workarounds have been proposed also with the scope to align Gadget forecasts to those from EwE and MSPM.

Model settings

Forward projections have been performed using the function *gadget.forward* from the R library Rgadget (Elvarsson 2016). Because we are interested in medium- and long-term projections we considered relevant to project recruitment based on stock-recruitment (S-R) relationships. In addition, recruitments of cod, herring and sprat in the Baltic are known to be highly affected by environmental variability (Cardinale et al. 2009, Bartolino et al. 2014). For this reason we performed projections by applying environmentally-sensitive S-R Ricker models for cod (1), herring (2) and sprat (3) modified from Margonski et al. (2010) as follows:

1. $R \sim \mu \cdot SSB \cdot RV_5 \cdot e^{-\alpha \cdot SSB}$
2. $R \sim \mu \cdot SSB \cdot SST_8 \cdot e^{-\alpha \cdot SSB}$
3. $R \sim \mu \cdot SSB \cdot SST_{78} \cdot e^{-\alpha \cdot SSB}$

where R is recruitment, SSB is spawning stock biomass, RV_5 is reproductive volume in May, and SST_{78} and SST_8 are sea surface temperature in August and averaged for July-August, respectively. Maturity ogive parameters applied in the projection were derived from 2000-2013 proportion of mature at age data from the WGBFAS assessment. *BAU* projections were performed by applying the last 5 years (2009-2013) average harvest rate by fleet.

Scenarios output

The following combinations of nutrient and climate scenario were used to force recruitment:

Scenario number	Nutrient scenario	Climate scenario	Climate model
1	BAU	PRS*	_
2	BAU	WAR1	ECHAM5
3	BAU	WAR2	HADCM3
4	BSAP	PRS*	_
5	BSAP	WAR1	ECHAM5
6	BSAP	WAR2	HADCM3

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Table 1.9. Combination of climate and nutrient scenarios explored under the BAU management scenario. Climate scenarios PRS (*not implemented in the rest of the caset study) corresponds to statistical present climate without warming, WAR1 and WAR2 are moderate and high warming, respectively. Nutrient scenarios BAU is business-as-usual and BSAP is the new Baltic Sea Action Plan.

Cod SSB projections are very similar until 2020 and depart sensibly for the period after. Trajectory for both the scenarios with moderate warming, and more under the BSAP nutrient scenario, increase rapidly until 2024-2025 to decrease during the last period. The scenario with no warming and BSAP nutrient load has a more progressive increase with the highest SSB in 2030. On the contrary, the other scenarios have a continuous decrease after 2020 which is most severe in the scenario with both high warming and high nutrient load. Herring SSB forecasts stabilize after few years, and cluster in two groups, with the two scenarios with high warming gaining the higher values. The same two high warming scenarios show also the highest values but a steady increase in the SSB of sprat (>0.8 x 10^6 tons), while the other scenarios stabilize quickly around 0.5-0.6 x 10^6 tons.

Overall, projected catches tend to mirror SSB patterns supporting the notion of a detrimental effect of high warming on cod, moderately positive effect on herring and markedly positive for sprat.

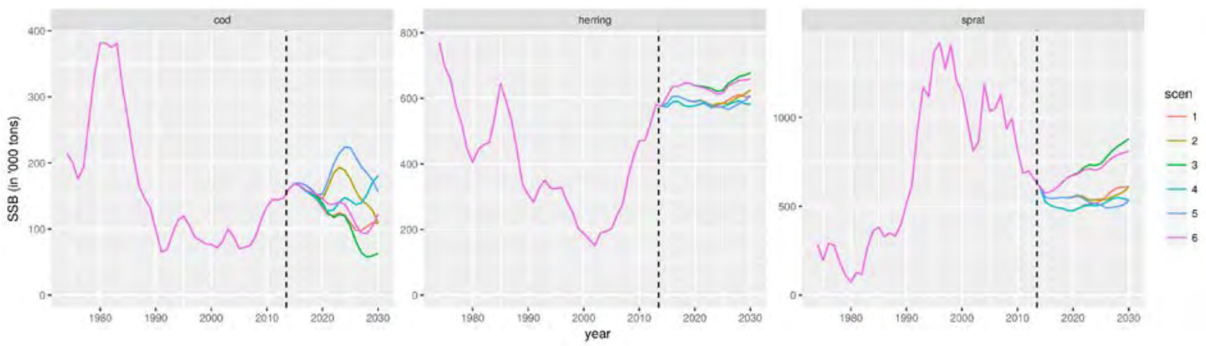


Figure 1.35. Time series of spawning stock biomass (SSB) for cod, herring and sprat with projections for BAU scenarios according to different climate and nutrient conditions as in table 1.9. Vertical dotted lines separate the historical (1974-2013) and the forecast (2014-2030) periods.

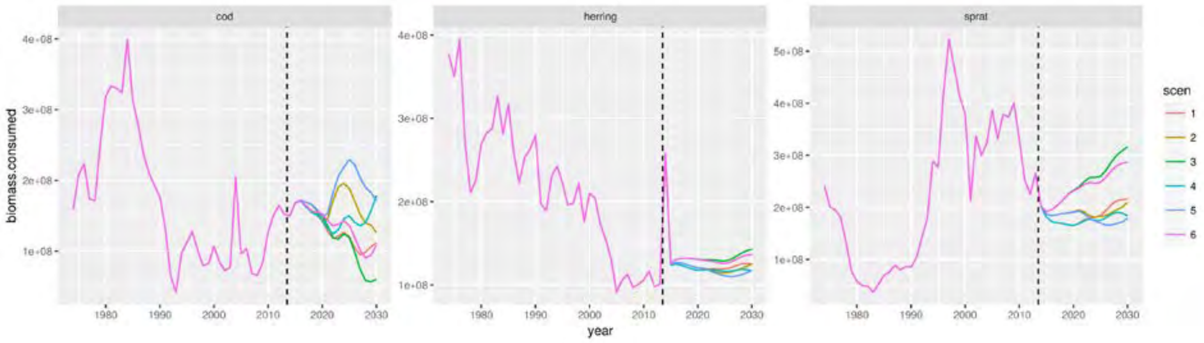


Figure 7.36. Time series of catch for cod, herring and sprat with projections for BAU scenarios according to different climate and nutrient conditions as in table 1.9. Vertical dotted lines separate the historical (1974-2013) and the forecast (2014-2030) periods.

Multispecies stock production model (MSPM)

The scenarios considered so far comprised all combinations of fishing mortality ranged from 0 to 1.4 with step of 0.1 for cod and clupeids. In the simulations fishing mortality of herring and sprat were assumed the same (to reduce total number of runs), giving in total 15*15=225 combinations of cod and clupeids fishing mortality. That allowed approximation of range of fishing mortalities associated with MSY for cod and clupeids as applied by ICES (BAU scenario) and scenarios MaxYieldCod. Other scenarios are not implemented yet, and an attempt for their implementation will be undertaken before version 2 of DST is released.

Model settings

The S-R models fitted to observed data and used in the prediction are presented in Fig. 1.37. The relation of observed recruitment to stock biomass is very week but the variance of cod and herring observed recruitment along fitted curves is relatively low, showing recruitment CV of 0.2-0.3 at given stock biomass. In case of sprat much higher variance is observed and recruitment CV at given biomass is about 0.7.

Predictions were performed till 2100, and growth rate was assumed as average of recent 5 years values.

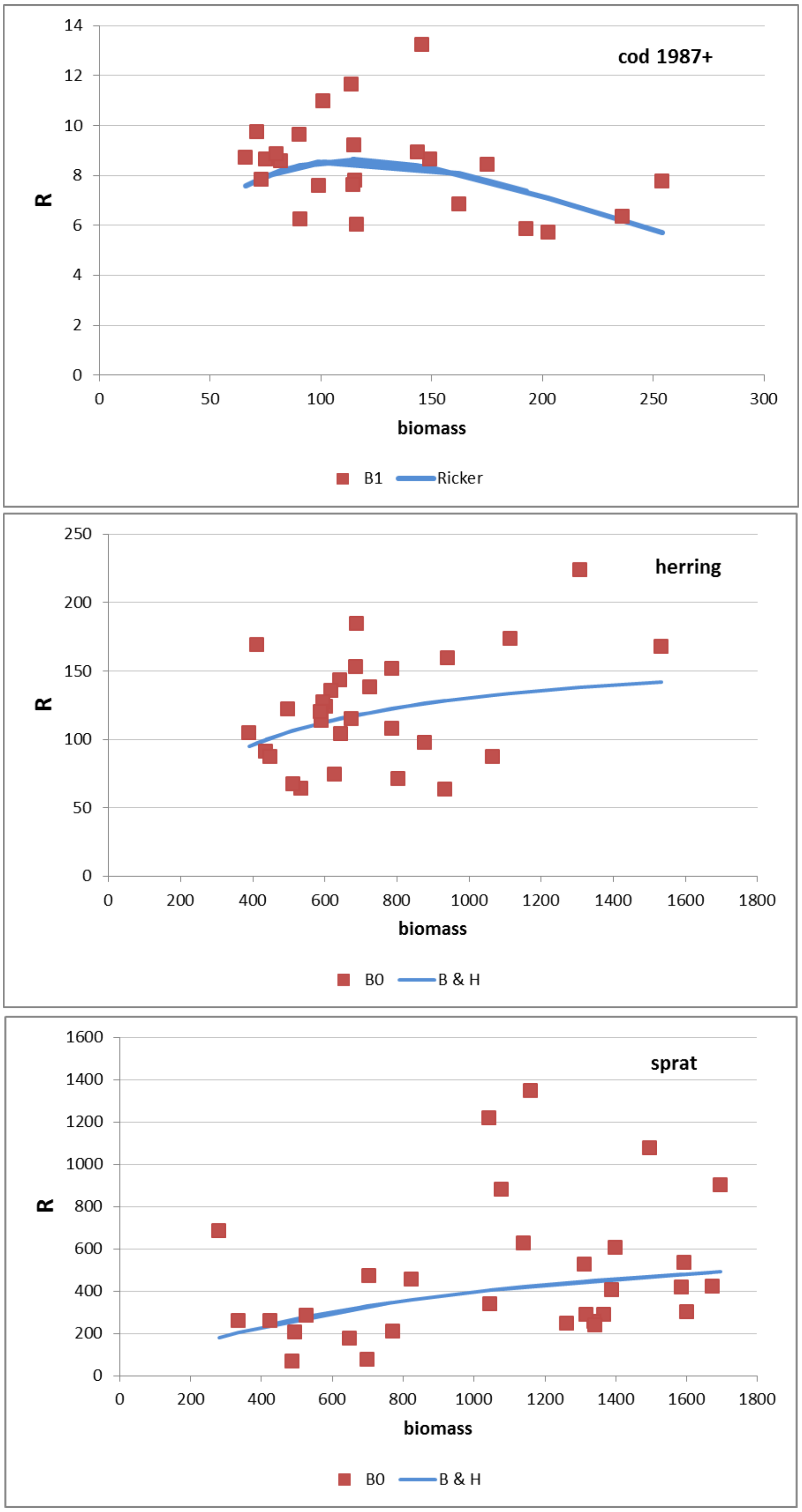
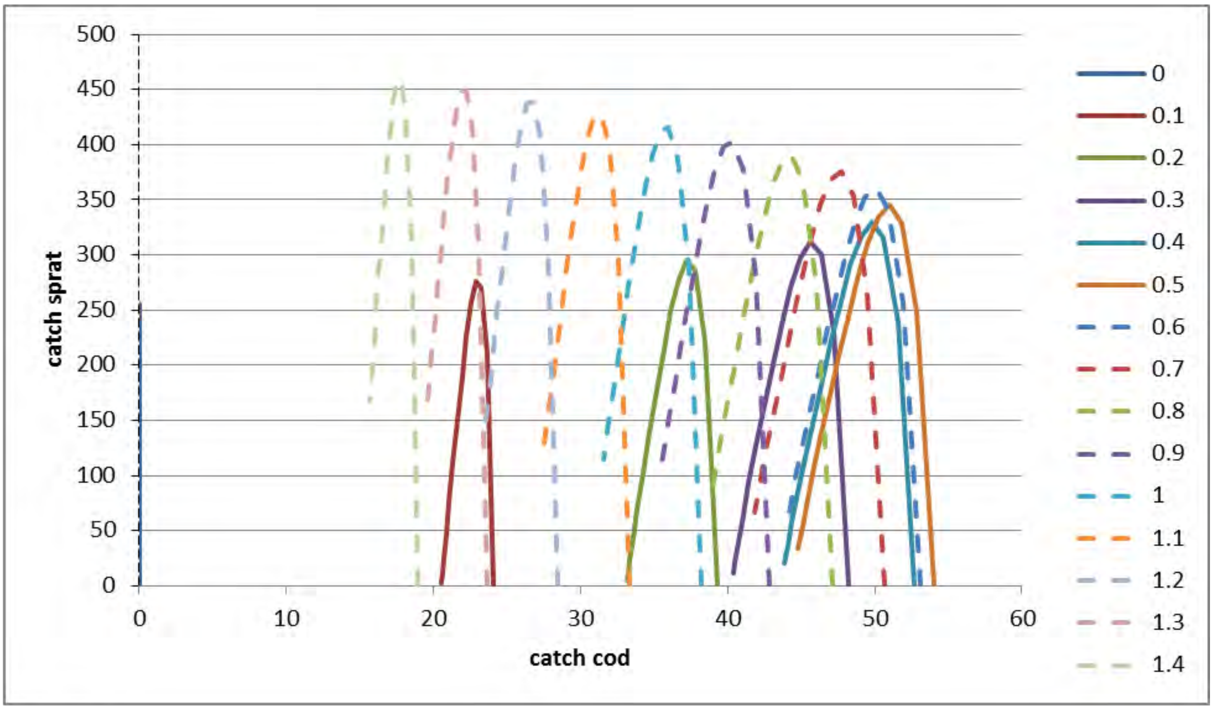


Fig. 1.37. Stock-recruitment relationship for cod (Ricker model), herring, and sprat (Beverton & Holt model) fitted using stock and recruitment estimates from MSPM. For recruitment biomass of age 1 is used for cod (B1) and biomass of age 0 for herring and sprat (B0)

Scenarios output

Deterministic results of multispecies prediction for range of Fs from 0 to 1.4 (step of 0.1) are presented in Fig. 1.38 as dependence of herring and sprat catches on cod caches at equilibrium. Figure legend shows cod fishing mortality and each “parabola like” curve represents situation for given F of cod and full range of herring and sprat Fs. Within given cod fishing mortality the cod catches decline slightly with increasing F of herring and sprat as with increasing fishing mortality of clupeids the amount of food for cod declines leading to stronger cannibalism and thus decline in cod biomass. Peaks of “parabolas” show MSY of herring or sprat under given fishing mortality of cod. Broken lines refer to cod fishing mortality higher than the F_{msy}. The Fmsy of cod equals about 0.48 and produces MSY of about 50 Kt (Fig. 1.39), with some range dependent on fishing mortality of clupeids.

Simulations indicate strong dependence of MSY of clupeids on fishing mortality applied to cod (Fig. 1.38). Given cod catches may be obtained with fishing mortality lower than the F_{msy} and F higher than F_{msy}, however, in the latter case the cod biomass may be markedly reduced, depending on how much the F_{msy} is exceeded. That would lead to lower predation pressure from cod to clupeids and thus much higher MSY of herring and sprat. The MSY of clupeids may differ almost by factor of 2.



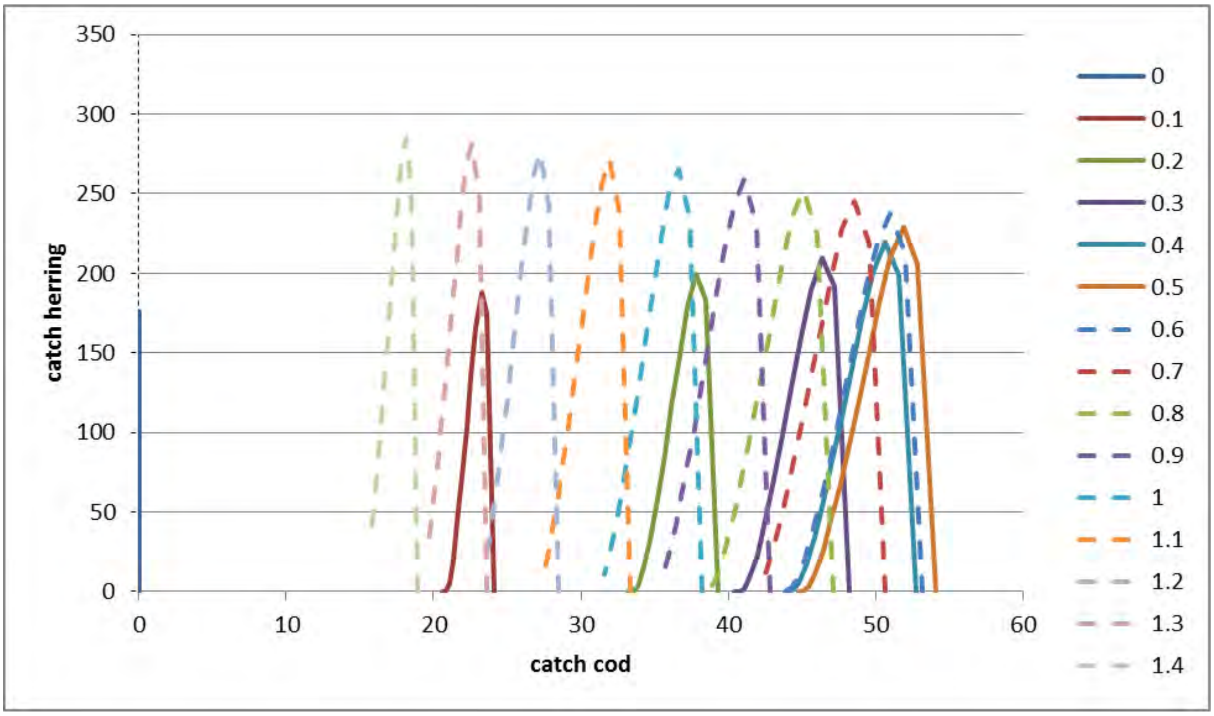


Fig. 1.38. The dependence of herring and sprat catches on cod caches. Each "parabola like" curve represents situation for given fishing mortality of cod and full range of herring and sprat Fs. Figure legend shows F for cod. Broken lines refer to cod fishing mortality higher than the Fmsy estimated within the MSPM.

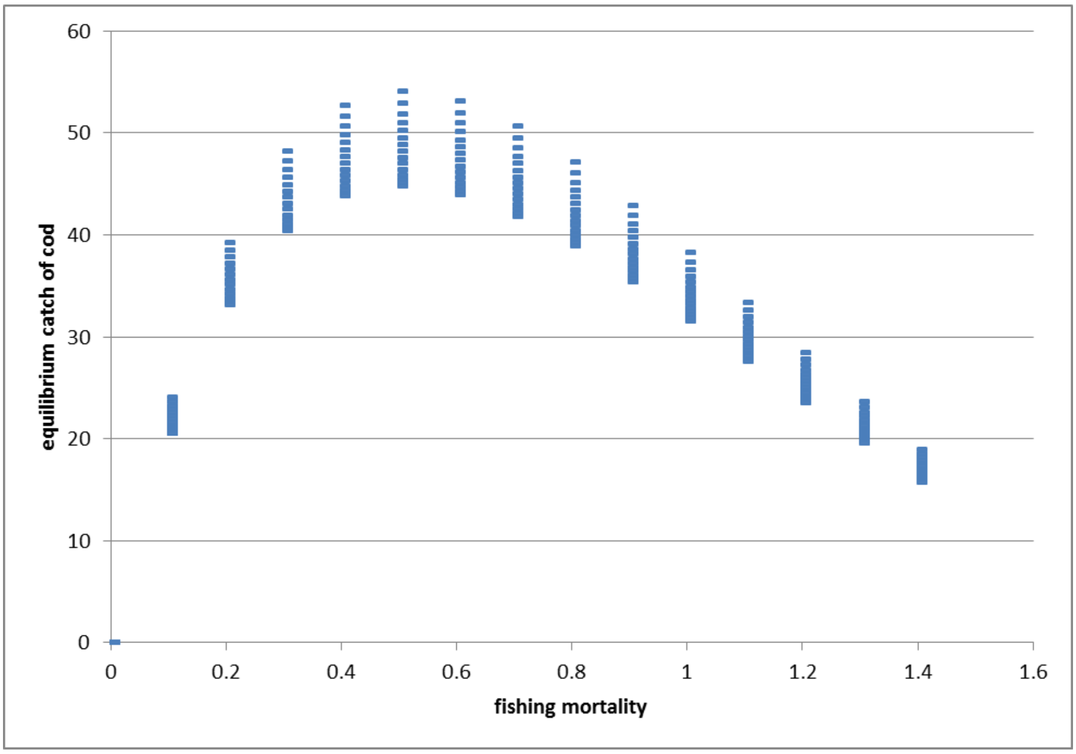


Fig. 1.39. Equilibrium catch of cod as dependent on fishing mortality. Ranges of points at given F refer to fishing mortality of clupeids (cod catch increases with declining clupeids F).

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2. North Sea case study

Brief description of the case study objectives

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

3. Need to achieve Fmsy
4. Landings Obligation
5. 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 Approach to Fisheries Management (EAFM) by developing:

- Novel data based on new tools and technologies
- Ecosystem models and assessment methods based on indicators of Good Environmental Status (GES)
- A Decision Support Framework (DSF) adapted to the needs of decision makers, managers, operators, and other stakeholders that will support the implementation of the new Common Fisheries Policy (CFP), Marine Strategy Framework Directive (MSFD) and Habitats Directive (HD)

However if possible, it also needed specific aspects such as compliance to be considered to address the Stakeholders' concerns. Reiterations of the model structure have been presented to stakeholders throughout the development phase by attending RAC meetings and arranging face to face and web-meetings.

Modelling framework

- The North Sea already had several working multispecies models as a result of earlier National and EU funded projects. These include the ICES SMS Model, (Lewy and Vinther 2004) the Le Mans Ensemble model (Thorpe et al 2015, Thorpe et al 2016) and an EwE model (Mackinson and Daskolov 2007) all of which have been reviewed by ICES WGSAM (ICES 2014, ICES 2015). In addition an existing form of the length based Charmingly Simple Model (Pope et al 2006) has been extended to a time varied version and is being further extended to deal with real species in addition to theoretical species characterised by their Loo and life history invariants. This model is needed both for estimating size based GES indicators and for including new (stable isotope) trophic level data. It is also useful for considering climate change. Multispecies Schaefer models and delay difference models are also being developed as cheap cheerful alternatives to more complex models and as means of describing the broad behaviour of the complex models. If possible a GADGET model will also be developed. However, the size and complexities of the North Sea may prove very computationally intensive for this form of model and consequently this model has a lower priority for development.
- Given the numbers of pre-existing models and developed and proposed models the initial modelling emphasis has been to develop the GREEN MODEL in a form that could be run using the outputs of the various available or new multispecies models of the North Sea in a coherent

fashion. This approach has various benefits, firstly it summarizes the behaviour of more complex models, secondly it allows common Fleet Constraint sub models, Social Economic sub models and GES sub models to be bolted on to the primary multispecies model in a consistent fashion. Even more importantly it provides a model that Stakeholder may use themselves to investigate trade-offs between management objectives, which could be used to readily address their additional concerns and which would support the decision support framework of MAREFRAME.

- An AMBER/RED model(s) are also being developed to consider area based concerns and to address the compliance issue in consultation with Stakeholders. The compliance issue is being addressed in collaboration with SAF21 consortium members and with Stakeholders.

Justification of why the modeling frameworks are suitable to address the objectives of the case study

Initially the Case study leader (John Pope NRC (Europe) Ltd proposed developing three models designated the GREEN, AMBER and RED Models that would be verified both by the inputs and comparisons with other models and by Stakeholder feedback. Figure 2.1 illustrates the models and this verification process.

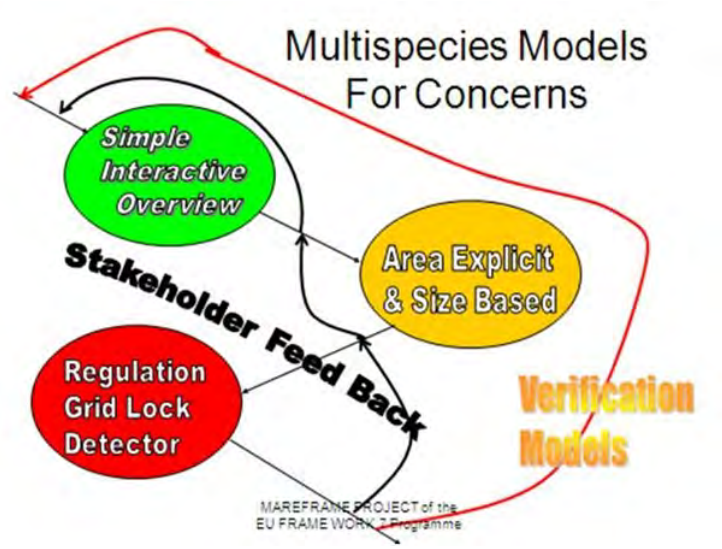

Figure 2.1. Schematic of the Modelling Strategy for the NS Case Study

The wide range of models considered represent varying hypotheses about the multispecies interactions of the North Sea and considering them all thus effectively addresses the issue of between model variations that is likely to dominate uncertainties in outcomes. Moreover all of the models have specific strengths that may add to the whole modelling process.

The GREEN MODEL in particular was designed for interaction with Stakeholders both individually and through the development of the DSF. In addition to their original choices the Stakeholders required additional outputs to be developed as model results became available to them. An example was they required measures of employment in on shore processing as well as at sea employment. They also asked for a measure of the changes in equity between different fleets for different management scenarios. In general stakeholders concerns are as far as possible built into this overview model. This avoids the need to duplicate this process for each individual model which would be both wasteful and extremely difficult where other people have developed the computer code. The main modules of the GREEN MODEL are depicted in Figure 2.2.

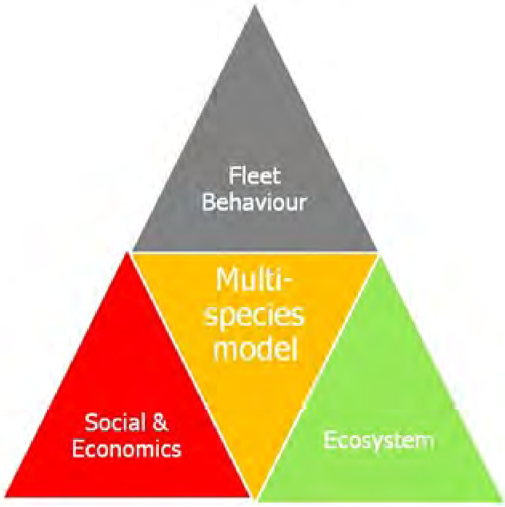

Figure 2.2. The Sub Modules of the GREEN MODEL.

The GREEN MODEL was initially fitted to the results of SMS (Lewy and Vinther 2004) which is an age based predation model that uses both the standard North Sea single species assessment data sets (catch at age, survey results and tuning series, weight and fecundity etc.) with additional comprehensive stomach content data from the main predators collected in 1981 and 1991. The GREEN MODEL also draws upon the results of the STECF fishing effort data set and the STECF Economic data sets (STECF 2016).

Brief description of the alternative models tried

The Le Mans model (Thorpe et al 2015, Thorpe et al 2016) is an Ensemble model that proposes a range of parameterizations and then retains only the subset of range of parameterizations that best predict North Sea historical stock trends. Initial comparisons suggest it shows rather weaker multispecies effects that are derived from SMS.

The available EwE status quo yield results are rather closer to those of SMS. However, these cannot be included in the full GREEN MODEL since EwE considers effort change by idealized fleets rather than by species fishing mortality rate. Comparative results for the steady state at status quo fishing mortality rates and at ±25% these rates can however be inserted into the Green Model to give the corresponding economic ,social and GES outputs.

Both Le Mans and EwE have the virtue of covering a wider range of species than the primary list of SMS and thus might be used to extend results to other species of interest to stakeholders (e.g. hake).

Schaefer multispecies models (Pope 1979) serve two roles in the MAREFRAME project. Firstly they are used to form the approximation to more complex models used in the GREEN MODEL (Pope 1989a, Pope 1989b, Collie et al 2006). This is essentially assuming that in the near field of fishing mortality rates the consequent changes in Yield, Discards and SSB may all be well described by a quadratic equation in fishing mortality rate for each species. Secondly, they provide a quick cheerful way to investigate if species interactions that SMS is not designed to consider may nevertheless exist. The possibility of the gadoid outburst in the North Sea resulting from the collapse of the herring (Cushing 1980) is an example of such an interaction that should be examined.

Simple Delay difference models (Pope 2003) may possibly fulfill this secondary role of the cheap cheerful model in a more satisfactory fashion since it is suspected that many of the multispecies interactions that occur in the North Sea do so by affecting realized levels of recruitment. (ICES 1987) Thus transitory equations might be better written in a delay difference form than as the logistic equation appropriate to the Schaefer model. Moreover, their structure admits of the inclusion of biological knowledge into their parameterization, e.g. growth rate and natural mortality rates. (Pope 2003)

The Charmingly Simple model was originally designed as a model of very low parameterization using species defined by their Loo alone (Pope et al 2006). It has been developed under MAREFRAME to consider transitory behaviors and is being developed further to provide a multispecies model with explicit species. Its low parameterization and size based structure makes it particularly suitable for considering size based GES indices. Moreover since it can be used as a trait based model it is suited to investigate climate change scenarios where displacement of species range to cooler (more northerly) areas may mean that the final North Sea fish species inventory might differ from those species that historically have inhabited the North Sea area. Such a transition in species would be difficult to handle in other models where species form an integral part of the structure.

The table below provides a brief synthesis of the models tried showing their purposes and their strengths and weaknesses where these are known.

Model	Advantages	Disadvantages
Frontend Model GREEN MODEL	Covers Biological, Technical interactions and provides Social and Economic and GES Outputs. It gives a direct measure of steady state results so is fast and can be rapidly optimized for scenarios. Biologically it has the advantages of whichever more complex model is approximated by its Schafer Model	The Schaefer approximation can get a little inaccurate at ±25% fishing mortality rate. It has the disadvantage that it does not provide a time series of future results but only the long term steady state. Biologically it has the disadvantages of whichever more complex model is approximated by its Schafer Model
Complex background model SMS	The SMS model considers predation between commercial species and can makes some allowance for predation by overlap species such as hake and mackerel and marine mammals. It is based upon main commercial species age structure. It uses full assessment data sets plus comprehensive stomach samples collected in 1981 and 1991. It considers detailed interactions that change with prey and predator size.	Only those feeding interactions that are captured in the 1981 or 1991 feeding sets are included and these are becoming rather dated. Potential interactions from pelagic species are not considered except via their role as alternative prey. It does not consider any changes in underlying productivity in the lower trophic levels.
Complex background model Le Mans	Allows wide range of hypotheses about interactions but weeds these down to those that are plausible based on historic time series. Covers more	There would seem to be a disconnect between feeding data and results. Not clear if it could consider climate change or not.

	species than SMS and allows consideration of model variation.	
Complex background model CSM	Length Based model originally conceived in terms of species defined by Loo groups and life history invariants. Very low number of parameters. Can be extended as a length based species multispecies model. Can consider climate change and novel data and good at considering length based GES indices.	Full species model still under development. The Proto-moment/Matrix Algebra basis of the transient model renders code very compact and efficient but very opaque.
Complex background model EwE	NS model available. Uses widely available soft-wear. Is an end to end model that can consider effects of changes to productivity. Gives results for more species than other models.	Existing NS model cannot be input to full GREEN MODEL because it is written in terms of fleets rather than species fishing mortality. Treatment of commercial species by EwE is rather rudimentary with respect to the key issue of size.
Cheap and Cheerful models. Multispecies Schaefer Model	Used as multispecies approximation model in the GREEN MODEL and generally approximates more complex models closely in the near field (±25%F'). Easy to fit to results of other models. Minimal parameterization. Handles stock recruitment as part of production.	Does not directly use biological data so it is difficult to fit the potential N(N+1) parameters by regression on past time series alone. Might not readily allow climate change to be considered.
Cheap and Cheerful models. Multispecies Delay difference Models	Allows more biological insight to be included than Schaefer models and can deal with real recruitment data in historic time series so should be easier to fit than a Schaefer MS model.	Model still under development. As with more complex models the critical stock recruitment relationships have to fitted.
Area explicit compliance models AMBER/RED models	Allows area based management and fleet decisions to be considered. Enables compliance to be considered.	Models still in development stage in collaboration with SAF21. Will need active Stakeholder involvement to parameterize compliance equations. Presently these models are only conceived as short term tactical models rather than long term models.

Brief summary on model selection.

North Sea Models have been compared elsewhere. Table 2 of D7.2.2 is the main means of comparing and contrasting models. Additionally, the Jacobian Matrix at the heart of providing Schaefer models that approximate the long term behavior also provides a clear route to making model comparisons. Our working hypothesis is to use the SMS results where they adequately describe the past but to consider additional interactions if these seem compelling. SMS has the virtue of having been reviewed repeatedly by the appropriate ICES expert group (ICES 2014, ICES 2016). The wide range of models considered will be used to develop a measure of our confidence in the resulting predictions. It is possible that the fully developed CSM may creep up on the inside rail to overtake SMS at the finishing post as first choice model to drive the Green Model approximation but this remains to be seen.

Best models (or subset of models used to inform the DST)

For now the GREEN MODEL based upon SMS results is used to provide the DSF scenario results. However, the nature of the GREEN MODEL makes it easy to provide these for any model whose long term behavior can be approximated by the GREEN MODEL since optimizing it to create the various DSF scenarios can then be done in a matter of minutes.

Parametrization and model fitting

Each of the models is or has been parameterized by fitting to historic time series. Therefore, typically catch (or where appropriate catch at age) are closely fitted along with either historic time series of abundance and/or biomass or for some of the cheap cheerful models by using the assessment results of single species or multispecies models to provide trends of historical abundance or biomass. Note that both single and multispecies models of North Sea stocks typically track trends in abundance and biomass rather closely and for some purposes it is more sensible to accept these trends that have been carefully considered by ICES expert groups rather than embarking on a difficult and perhaps contradictory tuning of simple models to raw data.

Scenarios evaluation

It is very simple to evaluate different scenarios in the GREEN MODEL using the SOLVER function of EXCEl to perform constrained optimizations of scenarios such as maximum economic yield. Typically these optimizations are easy to set up and take about 2 minutes to perform. The model provides tables of the standard criteria values needed for the DSF.

Description of scenarios

The Scenarios adopted in consultation with the Stakeholder for the DSF were as follows

1. Max Multispecies Economic Yield:- Profit is maximized for the North Sea fishery as a whole.
2. Max GVA – Gross Value Added:- Profit plus at sea labor costs is maximized for the North Sea fishery as a whole. This is a measure of the aggregate value of the fishery to society.

Max Landed value:- Landed Value is maximized for the North Sea fishery as a whole. Since the value of species differ widely this might be seen as the most sensible multispecies equivalent to MSY.

3. Max Conservation:- Fishing mortality rate is reduced to 75% of its status quo value for all species.
4. BAU – Buisness as usual:- Fishing mortality rate is kept at its status quo value for all species.
5. Max Economic Pelagic Yield:- As Scenario1 but only for the pelagic fishing gears
6. Max Economic Demersal Yield:- As Scenario 1 but only for the demersal fishing gears
7. Additional Scenarios have been considered in collaboration with Martin Pastoors and are likely to be introduced into the DSF.

NB: All scenarios are constrained to keep fishing mortality rate of all species in the range of 75% to 125% of their status quo values since this is the approximate range of safe extrapolation with the GREEN MODEL and in reality is probably as far as any multispecies model should be extrapolated since it is possible unstudied interactions might then begin to become important!

For each scenario the GREEN MODEL is programmed to provide a standard set of decision support criteria measures that were formulated with Stakeholder groups.

Model settings and assumptions to unable forward projections.

The adoption of the multispecies Schaefer model as the approximate multispecies model used to drive the GREEN MODEL allows prediction of any steady state that lies within the range 75% to 125% of status quo fishing mortality rates. This range was chosen to avoid extrapolating too far from current experience since wider changes might evoke unexpected interactions such as growth changes. This approximation is constructed from outputs from more complex models of yield, discards (where known) and SSB at the future steady state at status quo fishing, together with the equivalent long term steady states to be expected with a 10% increase in each individual species fishing mortality above status quo. This is the minimum information needed to solve parameters Ai and B(i, j) of the quadratic equation,

$Y'(i) = A(i)F'(i) + F'(i) * \Sigma_{\text{all j}} (B(i, j)*F'(j))$. Equation 2.1

Where Y'(i) is the steady state yield of species i, F'(i) Fishing mortality rate of species i written as the proportion of its status quo F and where both i and j=1:N where n is the number of species included in the ecosystem model.

For the most part the more complex ecosystem models are forward projection models so that providing these long term steady states only requires them to be projected into the long term future with their best current parameterization for the required range of F. A critical parameterization decision for all but the Schaefer model is the choice of stock recruitment relationship to apply for future recruitment. For the purpose of the quadratic approximation used (equation 2.1) it is helpful if stock recruitment relationships used are continuous and twice differentiable. Where this is not the case, notably with the widely used hockey stick Stock Recruitment relationship, some problems may arise. The existence of these problems can be detected by having estimates from the more complex model of long term steady states with each species status quo fishing mortality jointly and severally modified to 75%, 90% and 125%. This allow the adequacy of the Schaefer approximation to be judged. With SMS only the troubled whiting stock assessment differed appreciable at the extreme of this range and in fact the Schaefer approximation provided more believable results than the SMS prediction for this stock.

Detailed presentation of the best models selected

The model adopted to advise Stakeholders is the GREEN MODEL. Presently this is fitted to the detailed results of the SMS model that has the longest history of North Sea Multispecies models and which has been thoroughly reviewed by ICES WGSAM (ICES, 2014, ICES 2015).

Aspects of model implementation.

Figure 2.2 above shows the modular structure of the GREEN MODEL. Its central biological interaction (yellow) module is a multispecies Schaefer model based upon equation 2.1 above that predicts the long term consequences of changes to status quo fishing mortality rate on species landings, discards (where available) and SSB. The extent that fishing mortality rate on different species can change independently is controlled by the Fleet Behavior (Grey) module. This uses a download of the fishing effort data by gear type and country from the STECF data base (STECF 2016). Reductions in the fishing mortality on any species caught by any fleet have to be mirrored by equivalent reductions in the fishing mortality on the other species the fleet catches. However, some mitigation of other species fishing mortality being changed in lock step with the species with the greatest reduction (or smallest increase) is provided both

1. By providing a di minimus exemption for reducing mortality on those species that form an insignificant proportion of a fleets catch.
2. By allowing a percentage level of flexibility from the full reduction on fishing on other species that a fleet has to make when it must reduce fishing on a particular species.

Both the di minimus level and the level of flexibility can be modified by Stakeholders to take onboard their better understanding of these processes. This module gives the achievable as opposed to desired changes in fishing mortality that are consequent upon any management action. These results inform the central biological multispecies module so that it can provide estimates of landings, discards and SSB that result from the achievable changes in fishing mortality rate. The results also inform the Social and Economic (Red) module as to changes in the fishing effort and the changing share of each species landings for each Country and Fishing Gear fleet combination. In turn the Social and Economic module calculates the consequent changes in labor and non labor costs and the change in value of landings. The value of other catch is assumed to change prorate with fishing effort. Presently fish prices are treated as fixed by species but could easily be either preset by stakeholders or modified by price elasticity equations. Similarly labor and non labor costs are presently fixed but could be made modifiable by Stakeholders to consider more extreme scenarios. The species values and fleet linked costs adopted are based upon results from the STECF Economic data base and Economics report. A major problem in constructing this module was that these Economic data sets are based upon different fleet definitions and a broader area disaggregation than the STECF effort data. This means that a good deal of extrapolation and intelligent guess work had to be employed to use them for the North Sea. Given the resulting estimates of landings value and costs of each fleet we can then calculate the profit and GVA either in aggregate or broken down by fleet. Other measures of interest to Stakeholders such as changes in fish processor labor or changes in equity between fleet can also be calculated.

The final module is the Ecosystem (green) module that provides some indicators of Good Environmental Status (GES). The numbers of stocks below F_{MSY} (single species) or above the SSB reference points Blim, MSY Btrigger or Bmsy of each species at the long term steady state are passed from the Multispecies Module to this module as one such indicator . It also calculates some fishing mortality related GES indices. Presently these are:-

1. A measure of relative bottom disturbance (a weighted sum of fishing effort by species with emphasis on the flatfish fisheries that tend to have the largest bottom impact and away from pelagic gears that have none.
2. A measure of charismatic by-catch taken as the sum of fishing mortality of all fisheries. All effort is combined since it is difficult to apportion relative severities of bycatch to many and most may produce bycatch of charismatic species at certain times and locations.
3. A Measure of the Large Fish Index (LFI) . The bulk biomass MS Schaefer model used in the Multispecies module does not allow this metric to be calculated directly but studies based upon the CSM suggest that at long term steady states this and analogous measures are largely correlated to the different intensity of fishing mortality on fish with different Loo characteristics. These results can be used to interpret the effects of changes in the mortality rates on the different species included in the model and come up with an estimate of the likely change in the LFI or similar measures.

The GREEN MODEL is designed to be Stakeholder friendly, transportable, easy to understand and responsive. Consequently it is written in EXCEL which most Stakeholders own and know how to use. Its central Multispecies Module works as a simple Multispecies Schaefer model whose equilibrium under changed fishing mortality can be calculated by simple matrix algebra routines. This is very much quicker than it would be if results were calculated by a more complex model since these typically need to be iterated over many time steps to provide a new equilibrium. By contrast this model's response to a change in fishing mortality rate is virtually instantaneous. This is important since Stakeholders would be likely to be frustrated by slow responses. To simplify its use by Stakeholders fishing mortality rates are modified by moving sliders. Figure 2.3 shows the central control panel where these are modified.

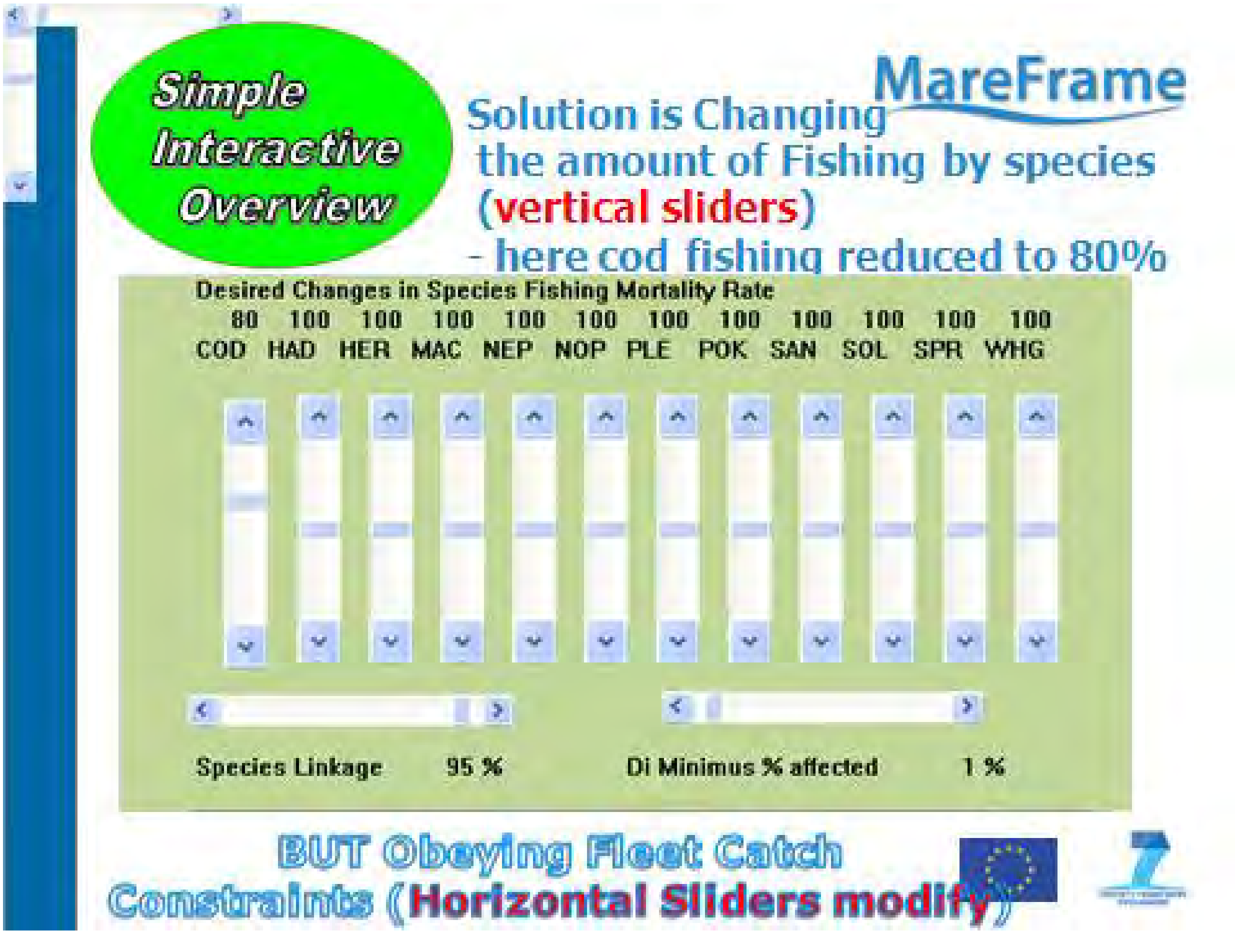

Figure 2.3 Control panel of the GREEN MODEL. The vertical sliders change the desired fishing

mortality by species. Here fishing mortality of cod is set to 80% of its status quo value with other species are left at their status quo fishing mortality rate. The two horizontal sliders set the parameters of how much flexibility exists within fleets to modify fishing mortality separately on different species. They thus govern the relationship between the desired and the achievable species fishing mortality rates.

Description of best models output

The first model output is the consequence of the choice of the di minimus level and the species linkage values. These, working together with the fleet catch by species data set, control the fishing mortality changes that must occur to all species caught by a fleet, to accommodate the largest reduction (or the smallest increase) in fishing mortality rate of any species it catches. Figure 2.4 shows a radar plot of the desired fishing mortality in blue and the achievable fishing mortality rate in red. Clearly to accommodate a reduction in cod fishing mortality(COD) to 80% of the status quo level requires fishing mortality to also be reduced upon haddock(HAD), whiting(WHG), sole(SOL), sandeel(SAN), saithe (POK), plaice(PLE) and Nephrops(NEP) but not on sprat(SPR), Norway pout(NOP), mackerel(MAC) or herring(HER) that are caught in distinct fisheries from cod.

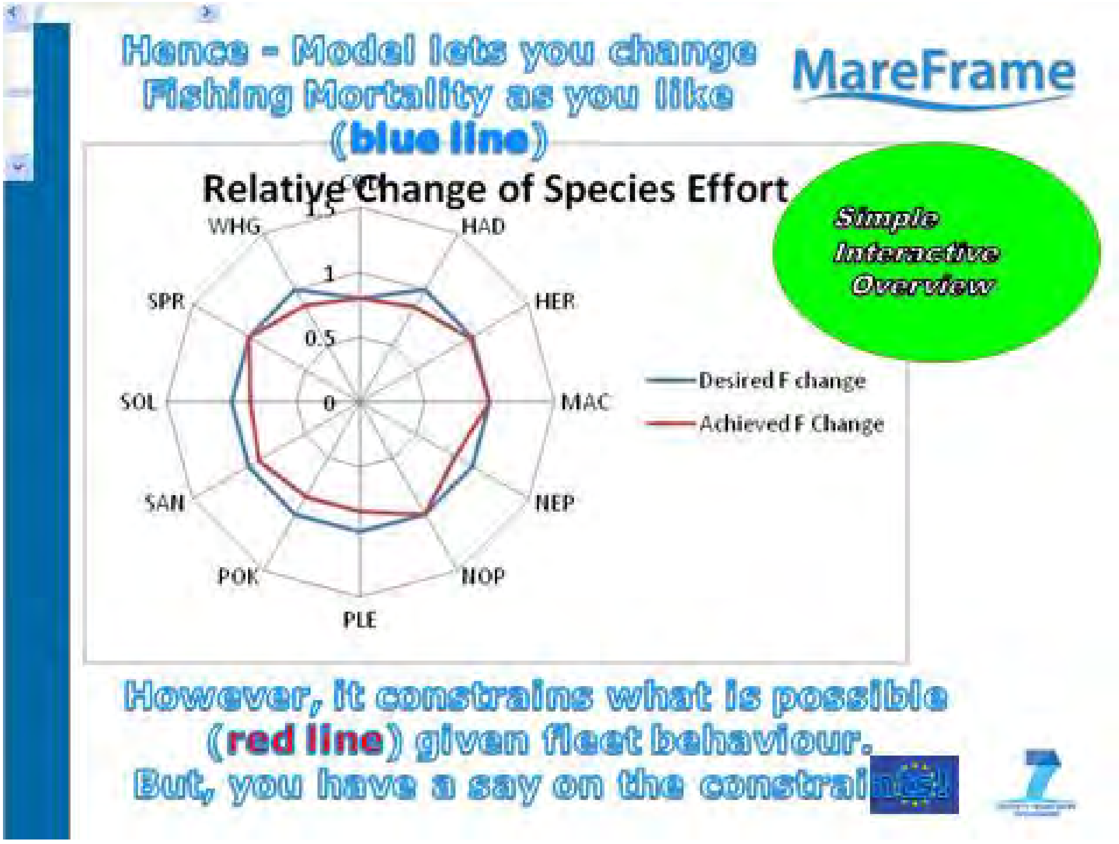

Figure 2.4 Results of desired and realized changes in fishing mortality rate.

Figure 2.5 shows the changes in landings to be expected consequent on the achievable changes in the fishing mortality. Note the changes in species catch are partly the results of the technical interactions causing reduced mortality on the species described above to accommodate the 20% reduction in cod fishing mortality given the fleet behavior. However some, for example the reduction in herring yield at the new steady state, result from the consequent increase in predator numbers, which results from reducing their fishing mortality rates. Similar results are provided for SSB and discards though the latter can only be calculated so far for cod, haddock, whiting and plaice for which comprehensive discard data are available. Note that this figure also shows 2 of the GES indices at the bottom.

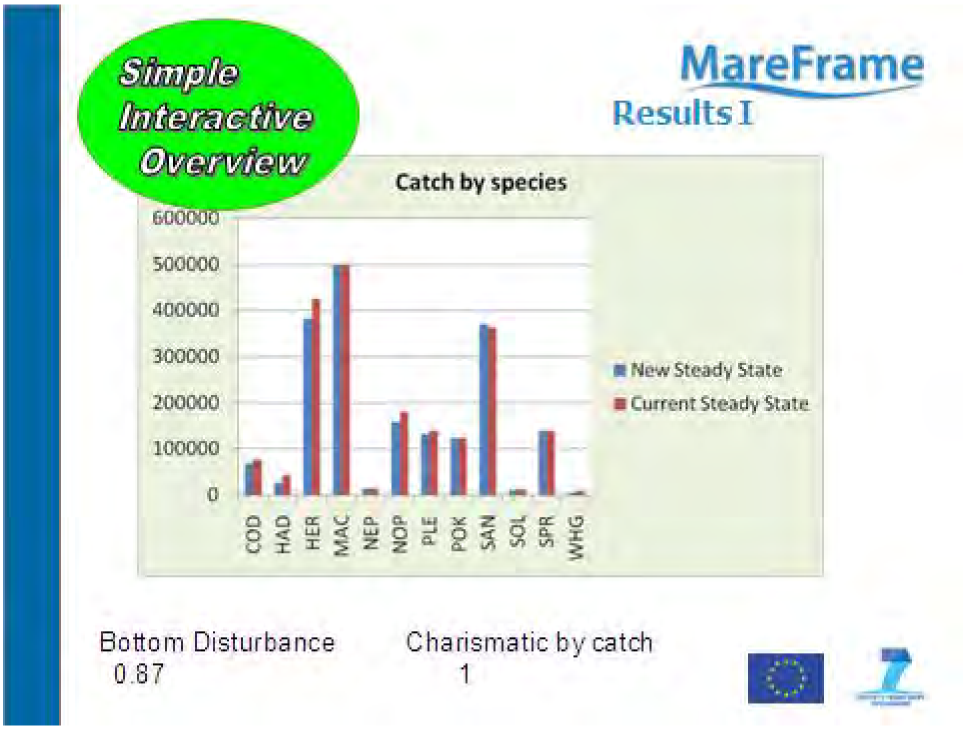

Figure 2.5 Status quo and new equilibrium landings estimates results for realized Fs.

In addition to these biological results the model also provides economic and social results. Figure 2.6 shows the main social and economic results provided both for status quo and the new equilibrium. Reducing fishing reduces the overall catch value and the labor and non-labor costs. The Gross Value added of the overall fishery declines but the profit is virtually unchanged (in fact it increases marginally but profit per remaining vessel (not shown) increases sharply. This later calculation assumes that effort is reduced by reducing the numbers of active fishing vessels rather than the existing vessel working fewer days.

The aggregate results shown in Figures 2.4 to 2.6 and equivalent figures for discards and SSB are presented alongside the control panel (Figure 2.3) rather in the format of a car dashboard to make it easy for Stakeholders to drive!

Additional tabs of the model provide similar results broken down by gear type or by Country. In principle they might be broken down by Country and Gear type fleets **but** the approximations needed to force the economic results into the gear behaviors data would probably mean such a disaggregation would be approaching the “Grain size” of the model.

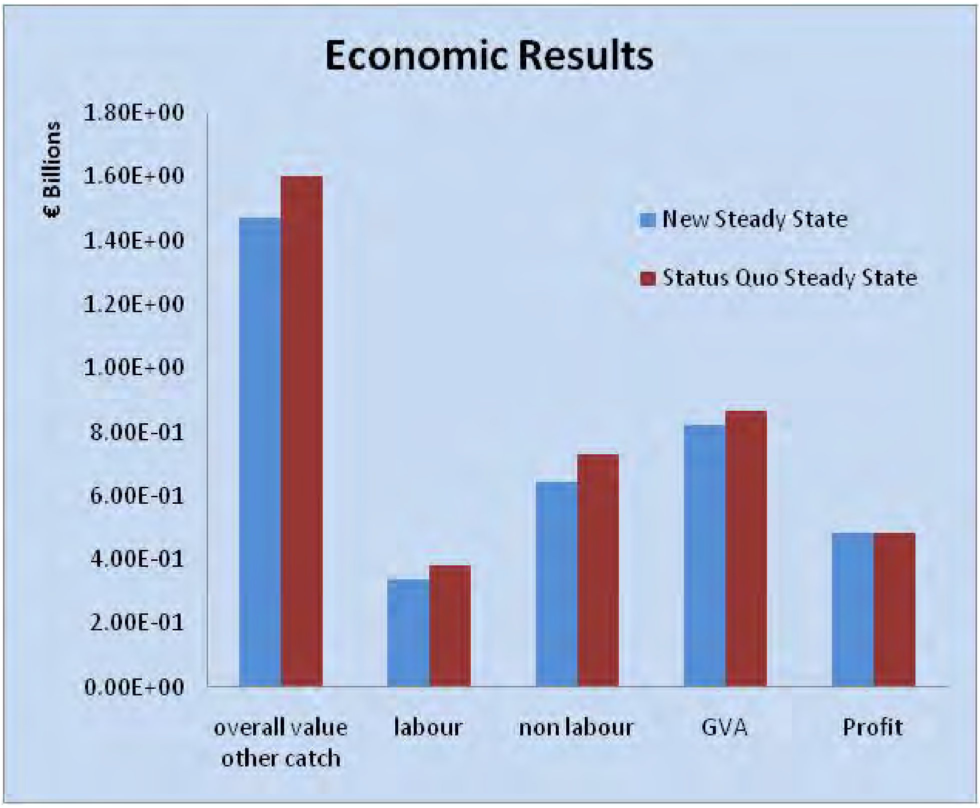

Figure 2.6 Economic results consequent upon wishing to changing cod fishing mortality to 80% of SQ fishing mortality rate.

In addition to being modifiable by the sliders, the fishing mortality rates on each species can also be modified by the EXCEL SOLVER optimization program to provide solutions to questions such as the fishing mortality rates corresponding to maximum achievable value or of maximum achievable overall profit or maximum achievable GVA. SOLVER allows robust constrained optimization and this has proved invaluable in developing the various DSF scenarios. For example Figure 2.7 shows the demersal species fishing mortality results from each of the DSF scenarios together with the single species Fmax value. It also shows the SSB of each species for each scenario relative to Blim. These were the criteria set by Stakeholder for the CFP compliance criteria used in decision support. The flexible form of the GREEN MODEL allows such scenarios to be rapidly developed or modified and standard output provided for DSF or model comparison tasks. Indeed one stakeholder was able to construct several potential new scenarios unaided in the course of a meeting!

In conclusion therefore the GREEN MODEL is a **Fast Flexible and Stakeholder Friendly model** that shows Stakeholders the linkages between trade-offs and which can also service the DSF needs of the North Sea case study for MAREFRAME.

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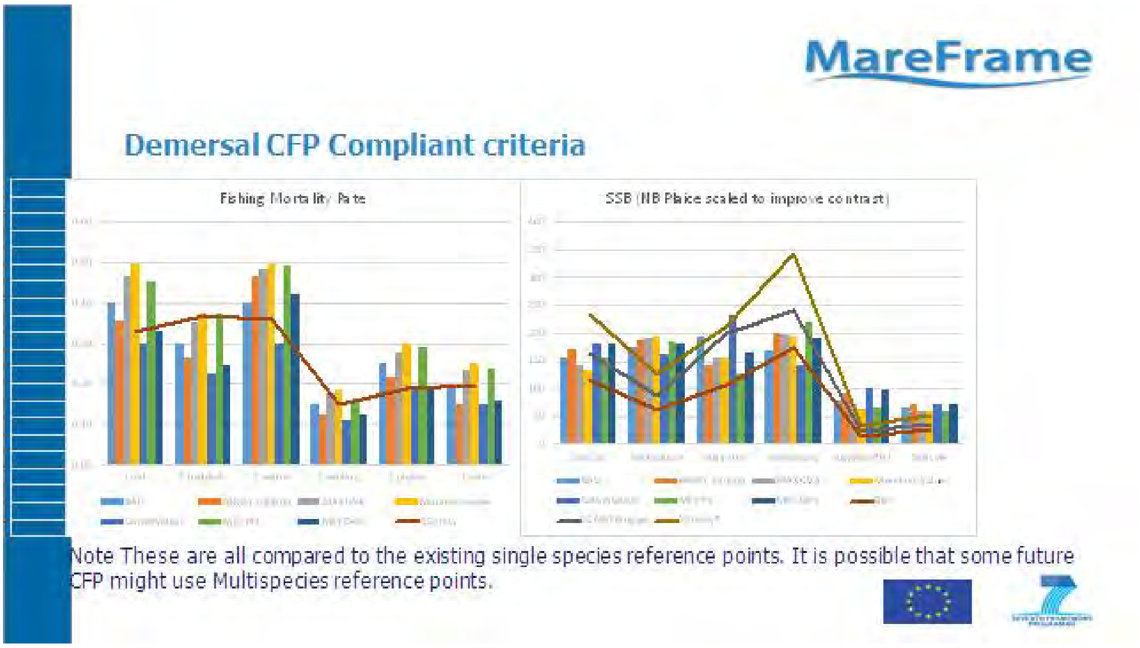

Figure 2.7. Each different DSF Scenario's level of fishing mortality for each demersal species compared to their Single species estimates of Fmax and their SSBs compared to their Blim.

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3. Iceland Waters case study

Brief description of the case study objectives

The stakeholders in the North Western Waters case-study have met on three separate occasions, where various aspects of the management of marine resources were discussed. In general, there is a good consensus within the stakeholder group with both the objectives and the implementation of the Icelandic Fisheries management act. The management act in essence sets the following goals:

- Strong and stable stocks
- Maintain biodiversity, food-web integrity, and sea-floor integrity
- Stable employment and settlement throughout Iceland
- Strong economic performance

Concerns were within the stakeholder group linked to the apparent uncertainty which frequent regulatory changes and relations to the fishing industry and the Icelandic community, at both local and national level. Other issues raised by the stakeholder group include removal of the quota consolidation barriers (currently 12% of TAC), effects of municipality controlled quota, aggregation of quotas in both the small (jig and line) and large type ITQ, and whether the industry should in general take socio-economic factors into account.

Modelling frameworks

The North Western Case-study implemented ecosystem models based on widely different approaches utilising two commonly used frameworks:

- **Gadget**: a statistical catch-at-length multi-species model
- **Atlantis**: a whole of ecosystem model

Gadget is shorthand for the "Globally applicable Area Dis-aggregated General Ecosystem Toolbox", which is a statistical model of marine ecosystems (previously known as BORMICON (Stefánsson and Pálsson, 1997) and Fleksibest). Gadget is an age-length structured forward-simulation model, coupled with an extensive set of data comparison and optimisation routines. Processes are generally modelled as dependent on length, but age is tracked in the models, and data can be compared on either a length and/or age scale. The model is designed as a multi-area and multi-fleet model capable of including predation and mixed fisheries issues, but it can also be used on a single species basis. The structure of the model is described in Begley and Howell (2004), and aformal mathematical description is given in Frøysa et al. (2002).

Gadget has been used extensively in the management of commercially exploited fish species in Icelandic waters, both as an auxiliary model (e.g. órarson and Elvarsson 2013) and as the basis for advice for many species including golden redfish and ling (see Thordarson, Elvarsson, and Kristinsson 2011). One of the main reasons for its use has been its ability to work with diverse sets of data and varying degree of data completeness. To aid in the model development and subsequent analysis a specialised R-package, Rgadget, was developed for the use in this case-study.

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 addition work on a model based on an Ecopath with Ecosim model is in its initial phase.

Conceptual model

The Icelandic continental shelf is positioned at a dynamic frontier of the Atlantic current and the artic circle, where an influx of warmer water meets the colder Artic waters Astthorsson, Gislason, and Jonsson (2007). As such it features a complicated ecosystem and, as in many other areas in the North Atlantic, gadoid species, especially cod, are the main focus of the fishery. Species interaction in Icelandic waters has received considerable attention (e.g. Gislason and S. Ástórsson 1997, Stefánsson and Pálsson (1997)).

Historically the location of this dynamic front has been seen to vary considerably. On a longer time scale, during the small ice age, the frontier shifted further south which in turn shifted the distribution of several fish species further south. At that time contemporary descriptions indicated that little or no fish were caught by local fishermen Jónsson (1994).

On a smaller timescale small scale variations in the spatial distribution of commercially important species have been observed. The great herring collapse of the 1960s is an extreme case (potentially due to overharvesting or changes in the ecosystem). Capelin migrations have been seen to vary according to temperature Ó. K. Pálsson et al. (2014) and a north south migration has been observed in many species. Concomitant with these changes, fluctuations have been observed in the average sea temperatures.

Exploitation of marine resources in Icelandic waters is fairly diverse, ranging from deep water fisheries s.a. golden redfish and Greenland halibut, to gadoid species e.g. cod and haddock, pelagic fish species and marine mammals. Through stakeholder interaction it was revealed that their primary concern in maintaining a strong and stable cod fishery, other priorities were auxiliary. Therefore the main focus of the modeling effort using Gadget was on cod and the species caught in tandem, namely haddock and saithe. Three other species, ling, tusk and wolffish were included in the model as well as they are caught is the same fishery. Figure 1 shows an overview of the locations of the catches of the species in the fishery.

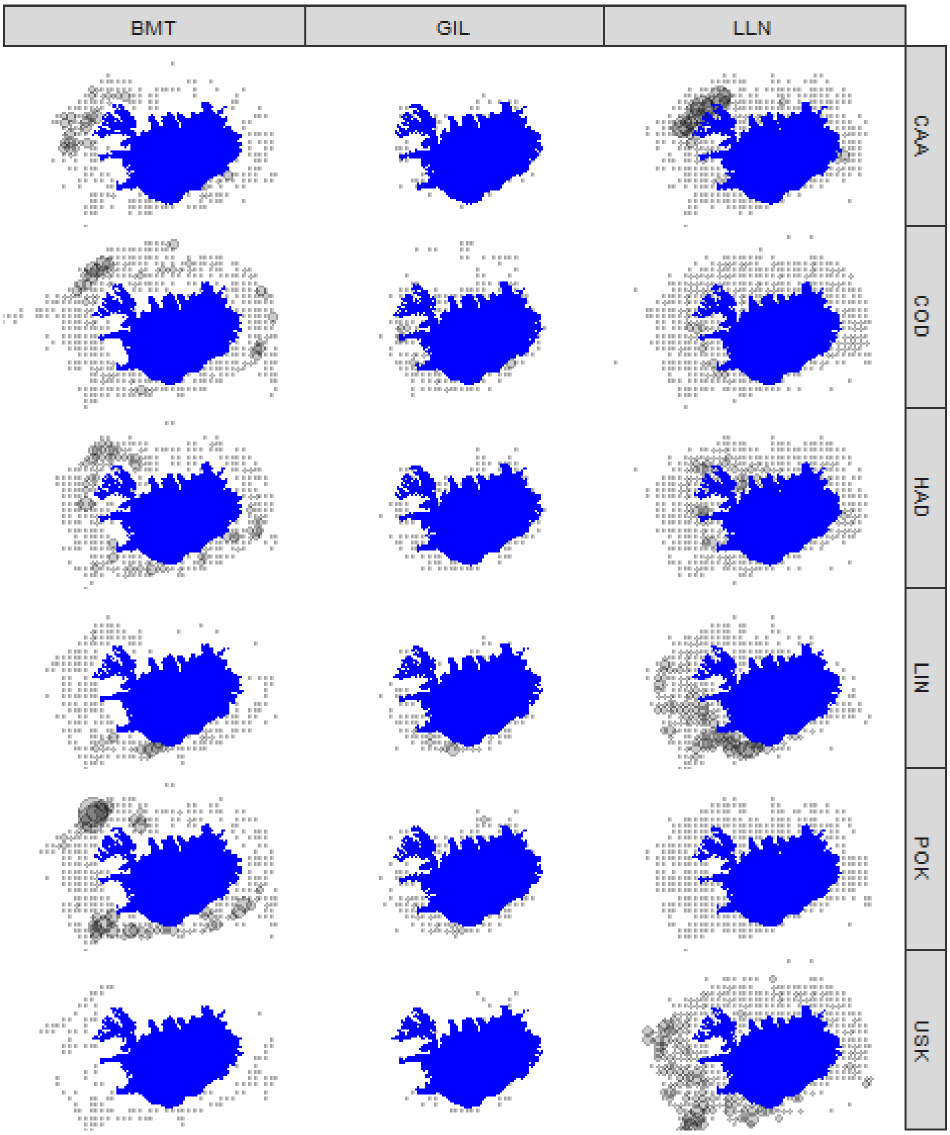

Fig. 3.1: Species occurrance in the fishery by gear type

When looking at the fleet composition it can be observed that considerable changes have occurred in recent years, as illustrated in figure 3.2. Notably the bottom trawl effort has decreased considerably.

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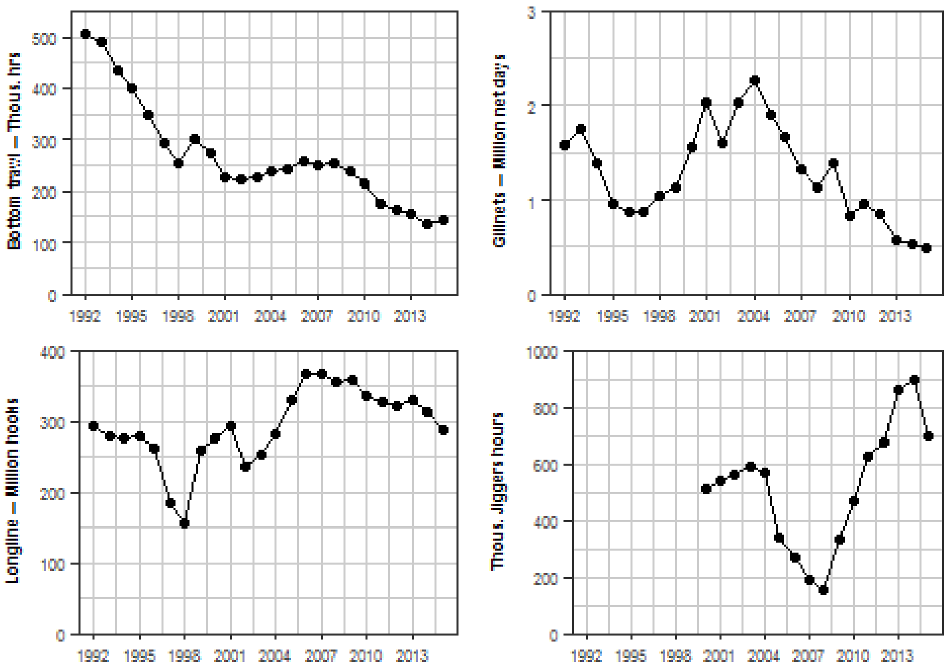

Fig. 3.2: Effort as a function of year by fleet segment

Due to these recent changes in fleet operations and differences in species catch proportions by gear the general setup of the model implemented in Gadget is as illustrated in figure 3.3. Three main commercial fleets are implemented within the model, bottom trawlers, gillnetters and longliners. Jiggers are assumed to behave within the model as longliners.

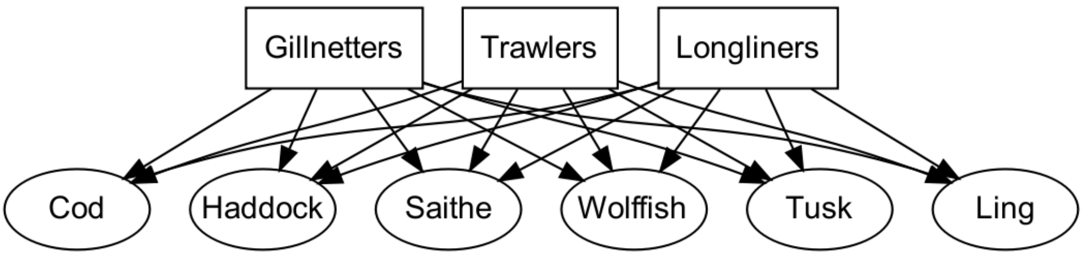

Fig. 3.3: Schematic overview of the Gadget model for the North Western Waters case study

Brief description of the alternative models tried and summary on model selection wherever possible

Slight variations of the Gadget models were considered, notably a model where deep water species, such as golden redfish, were included and another where minke whales and cod interactions. These model variants including deep water species was considered to be outside of the scope of the case-study.

Model results

The primary model for the case-study was built using the Gadget framework. It utilises available data from 1960 to 2015. In particular the data includes:

- Species landings by quarter since 1962
- Catch at age prior to 1984 for cod, haddock and saithe
- Port samples from the commercial operations. This include measurements of age and length.
- Two survey series, from the Icelandic spring groundfish survey which started in 1984 and the Icelandic Autumn trawl survey that started in 1996.
    - These surveys further supply biological measurements of age, length and maturity

Fitting procedure

The model is contrasted against all these datasets using a weigthed likelihood function and the model parameters are estimated by mininmizing this weighted likelihood function. Gadget's function minimizer, based on the negative log--likelihood, varies the model parameters, runs a full simulation, and calculates a new output. This process is repeated until a minimum is obtained. The model has three alternative optimising algorithms linked to it, a wide area search *simulated annealing* (Corana et al. 1987), a local search *Hooke and Jeeves algorithm* (Hooke and Jeeves 1961) and finally one based on the Broyden-Fletcher-Goldfarb-Shanno algorithm hereafter termed BFGS.

The simulated annealing and Hooke-Jeeves algorithms are not gradient based, and there is therefore no requirement fir the likelihood surface to be smooth. Consequently neither of the two algorithms returns estimates of the Hessian matrix. Simulated annealing is more robust than Hooke and Jeeves and can find a global optima where there are multiple optima but needs about 2-3 times the order of magnitude number of iterations than the Hooke and Jeeves algorithm.

BFGS is a quasi-Newton optimisation method that uses information about the gradient of the function at the current point to calculate the best direction to look for a better point. Using this information the BFGS algorithm can iteratively calculate a better approximation to the inverse Hessian matrix. When compared to the two other algorithms implemented in Gadget, BFGS is a very local search compared to simulated annealing and is more computationally intensive than the Hooke and Jeeves. However the gradient search in BFGS is more accurate than the step-wise search of Hooke and Jeeves and may therefore give a more accurate estimation of the optimum. The BFGS algorithm used in Gadget is derived from that presented by Bertsekas (1999).

Assessment results

To illustrate the model stock assessment the standard performance plots were created, as illustrated in figure 3.4. The model illustrates well recent changes in fleet efforts, as the fishing mortality has been reduced substantially in the past 10 years. This is inline with the observed changes in the bottom trawl effort.

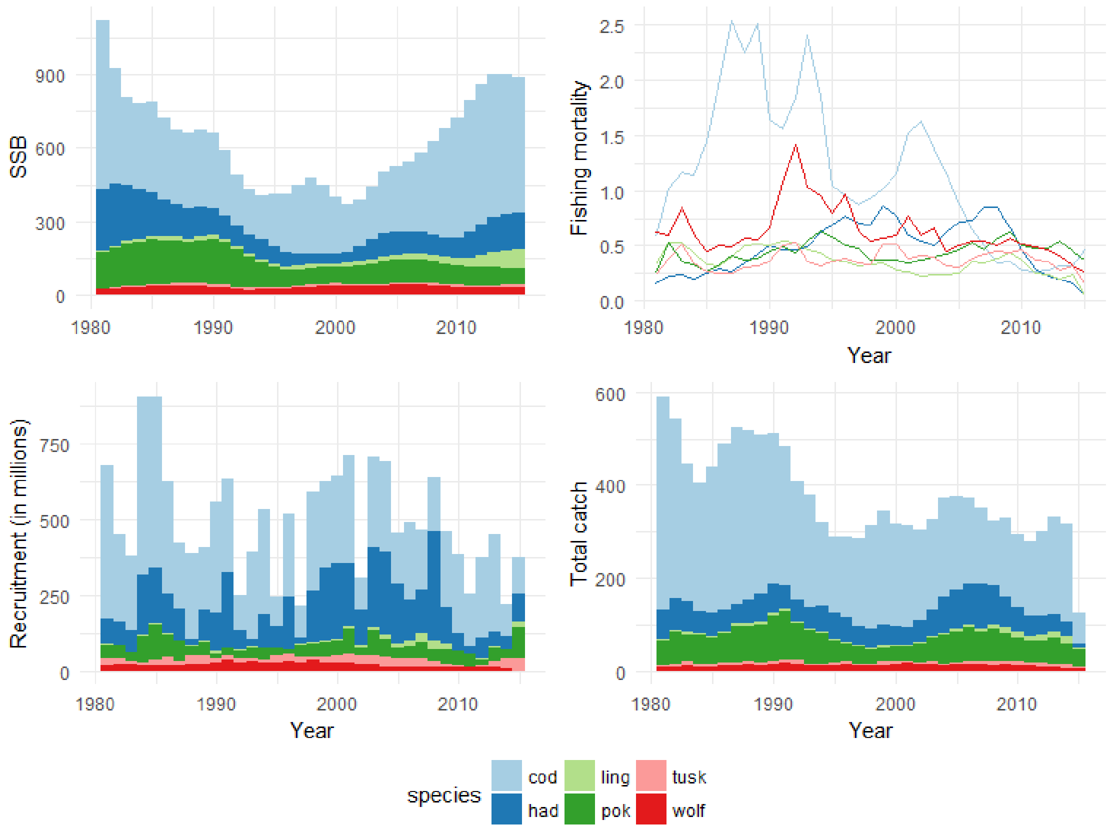

Fig. 3.4: Assessment overview of stock status by year. The top left panel shows the estimated biomass , top right the fishing mortality, bottom left recruitment and bottom right catches.

Scenarios evaluation

Description of scenarios

- **Business as usual**: This scenario serves as baseline to other potential management scenarios. In the scenario the current status of management is maintained and the effects on the status of the ecosystem explored going forward. In terms of control variables this entails that the current fleet composition and harvest rate maintained.
- **Cod to Fmsy**: This scenario offers a slight modification of scenario 1 as here the harvest rate is adjusted in such a way that the yield of the cod fishery reaches its maximum while fleet composition remains fixed.
- **Changes in fleet composition**: The effects of specific changes in fleet composition in terms of management restrictions are explored. Currently the small scale fishery is allotted a proportion of the quota that cannot be transferred to larger fishing vessels. This scenario analyses of the effects of removing this restriction on quota transfer from the small scale fishery.
- **Multi-species maximum sustainable/economic yield**: The fishing rate and fleet composition is altered such that either of the following yield levels is attained:
    - Maximum sustainable yield from the resource
    - Maximum economic yield

- **Environmental concerns**: This scenario investigates the effects of adjusting the harvest rate and fleet composition is such a way that overfishing and over depletion is prevented and the effects on the environment such as CO2 emissions and damage to the sea floor is reduced.

Model settings and assumptions to enable forward projections

The stock status for the different species was projected forward based on simple assumptions on process error. Stochastic simulations, as implemented Elvarsson and órarson (2014), were implemented using an autoregressive form:

$$\overline{R}_y = \overline{R}_{y-1} + \epsilon_y$$

where ϵ_y is a mean zero gaussian with variance σ^2. This is of course a crude approximation to the recruitment process that assumes that recruitment is not impaired during the projection period. This form of the process error was implemented for all species. Fishing operations in the forward projections were parametrised in terms of harvest rates, i.e. ratio of the biomass available to the fishery. This process accounts for differences in the fleet selectivities.

Ecosystem model outputs on each scenario

To illustrate the effects of each scenario, long term average statistics were derived based the forward projections described above. Figures 3.5 and 3.6 illustrate the steady state SSB and yield in tonnes when keeping the ratio between the fleets fixed. From these figures it is apparent that, compared with BAU scenario, cod to F_{msy} would mean a 20% increase in overall effort. However this is not true for other species. Saithe is currently far above its single species F_{msy} and taking cod to F_{msy} would result harvest rates exceeding precautionary reference points. Other species, i.e. ling, tusk and wolffish, would be classified to be overfished while above B_{pa}.

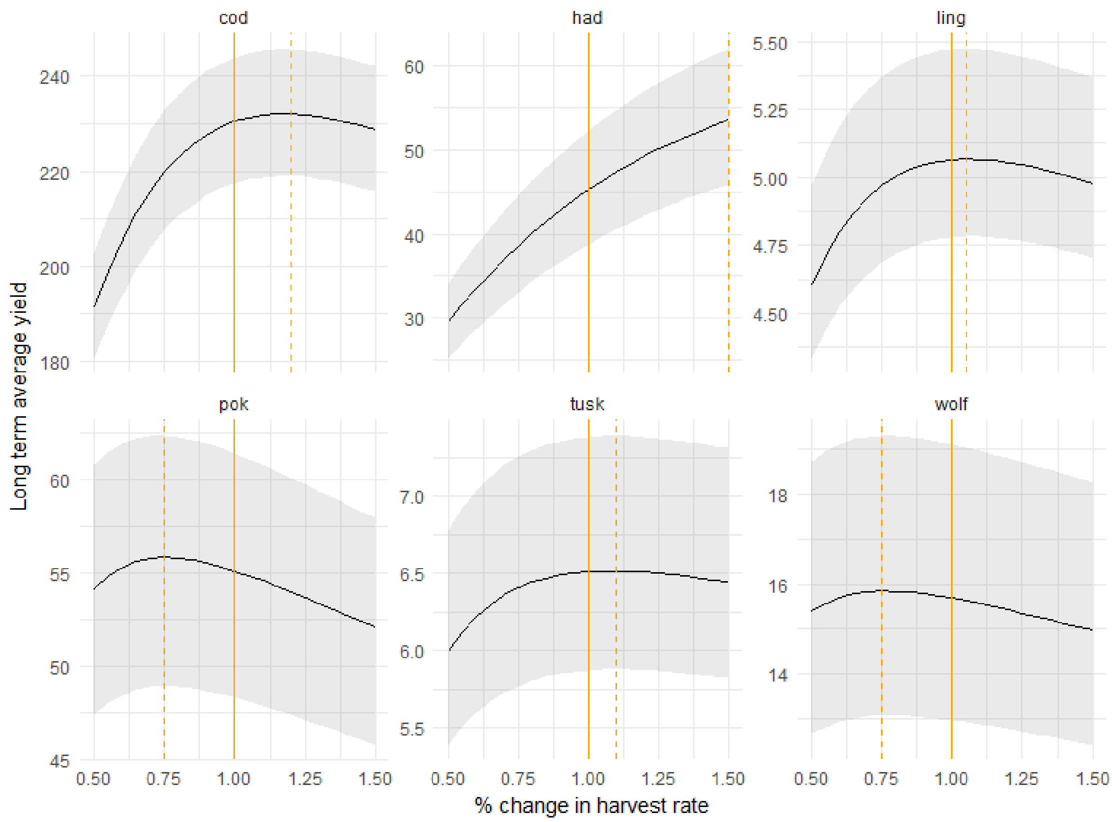

Fig 3.5: Long term average single species yield as function of relative harvest rate keeping the fleet ratios fixed. Solid black line indicates the estimated average while the shaded region the 95% confidence intervals. Vertical orange lines indicate status quo (solid lines) and optimal (dashed lines) harvest rates.

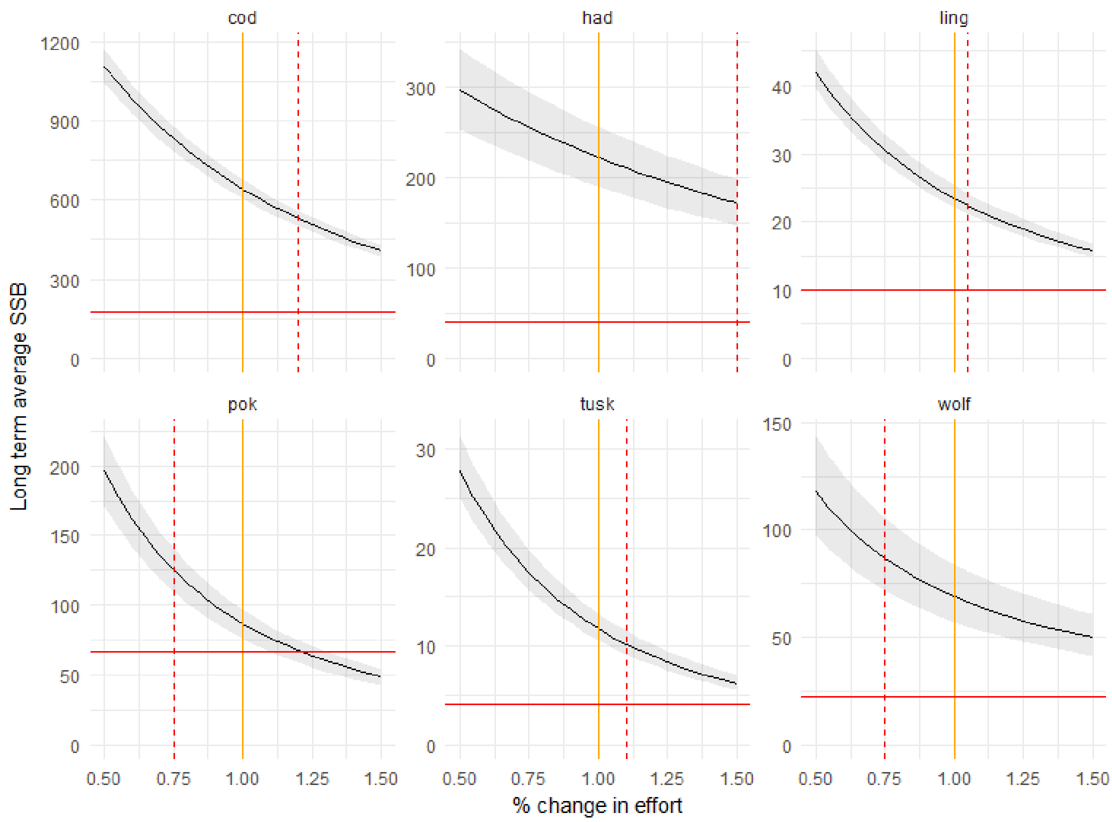

Fig. 3.6: Long term average SSB as function of relative harvest rate keeping fleet ratios fixed. Solid black line indicates the estimated average while the shaded region the 95% confidence intervals. Vertical orange lines indicate status quo (solid lines) and optimal (dashed lines) harvest rates, and horizontal line the B_{pa}.

Maximum combined yield is illustrated in figure 3.7. They indicate that overall very little changes in yield and profit is expected when varying the fleet ratio. However, B_{pa} for saithe will move further to the right suggesting that reducing the bottom trawl effort would decrease the fishing pressure on saithe while increasing the catches of other species. When looking at the economic yield is is notable that nearly 25% decrease in harvest rate would maximise the profit.

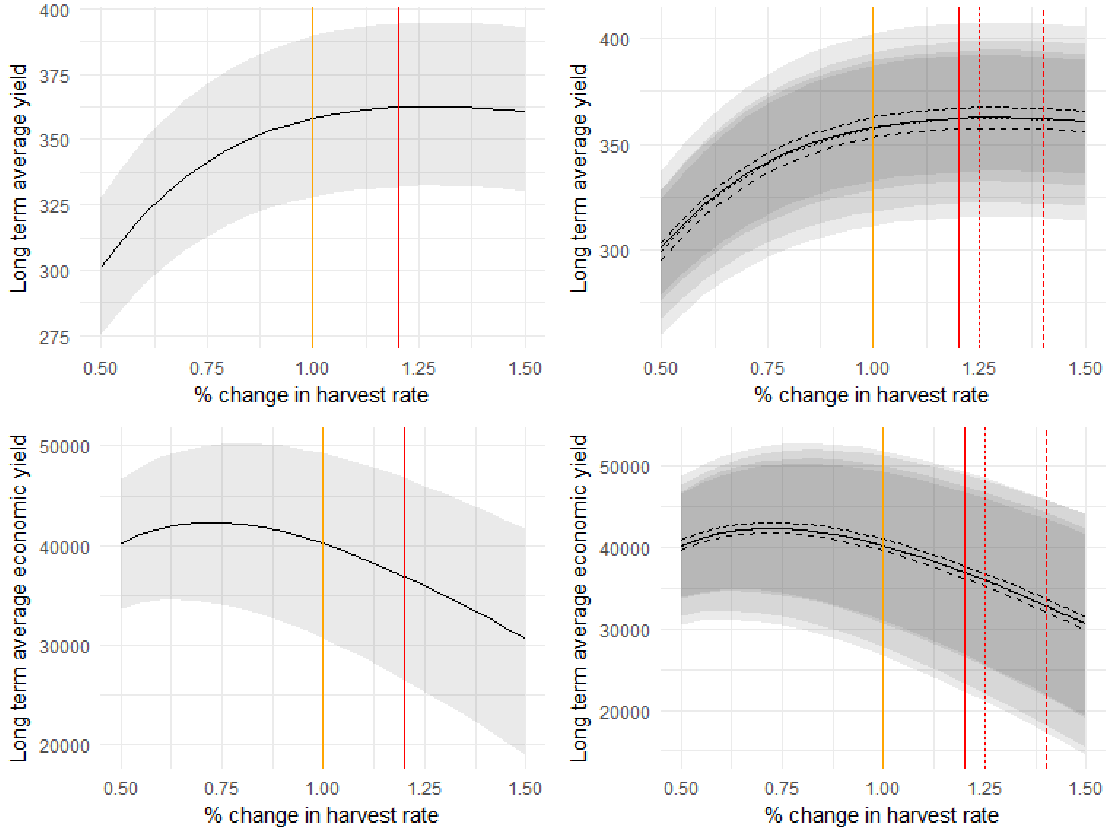

Fig 3.7: Long term average (economic) yield as function of relative harvest rate. Solid black line indicates the estimated average while the shaded region the 95% confidence intervals. Broken black lines inidicate varying proportions longlines in the total fleet effort relative to bottom trawlers. Vertical orange lines indicate status quo (solid lines) and H_{pa} (dashed red lines) i.e. precautionary limit harvest rates as a function of longliner effort ratio.

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4. West Scotland case study

Brief description of the case study objectives

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 (Fig. 4.1). 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, which may have a negativale impact on gadoid stocks (SCOS, 2008). The increase in grey seals population has recently been linked to an increase of predation mortality on cod which could jeopardise effort to recover the stock (Cook et al., 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. Lastly, GES must be achieved by 2020 as per specified by the MSFD. 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 use of ecosystem models allows for investigating these management issues whilst accounting for prey-predator interactions as well as assessing the impact of the whole ecosystem.

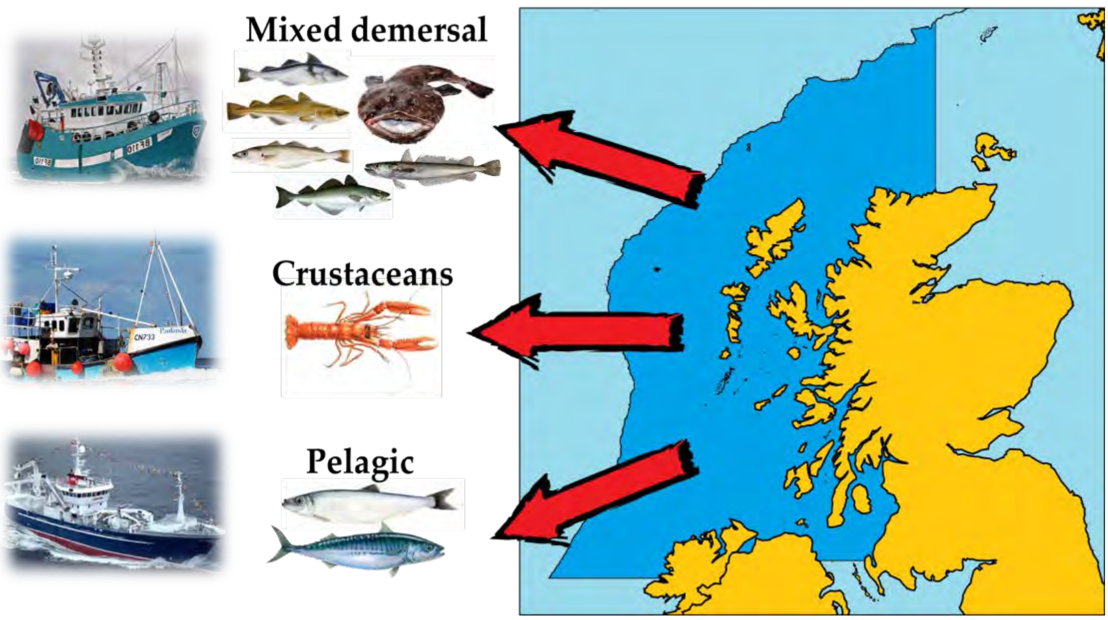

Figure 4.1. The area considered in the west of Scotland case study (dark blue) and the three main fisheries in the area

MareFrame researchers met with stakeholders from the west of Scotland case study (NWWAC, SFF and SWFPA) at the start of the project (May 2014). It was mutually agreed that the ecosystem models would be used to explore alternative management strategies co-designed with stakeholders to address the following research questions:

- What are the management measure(s) required to recover the stocks of cod and whiting?
- What is the optimum fleet effort distribution between demersal, Nephrops and pelagic trawls?
- What is the impact of seal predation?

Once the first of the two model employed in the west of Scotland case study was parameterised and ready to use, MareFrame researchers met with stakeholders again to agree on a set of management alternatives designed to address the issues listed above which were to be simulated with the model. These alternatives are:

- Status quo F: keeping the fishing mortality (F) constant and equal to the last historical year (2013) for all species
- Current path: apply the ICES single species F_{MSY} for species for which this value is defined by ICES (see table 4.1 for values). When values were not available for the model area, best available estimates (e.g. neighbouring areas) were used. Species with no F_{MSY} values are harvested at status quo.
- Nephrops and pelagic: same as current path, but with F increased by 20 and 30% for Nephrops and pelagic fish species respectively (the % of increase was identified through preliminary analyses)
- Seal cull: same as current path, but with a cull of 10% of the seal population every year
- Cod recovery: same as current path, but with cod fished at F=0.05 (value corresponding to residual F when the stock is not targeted, based on historical values for whiting which is not targeted since 2006) instead of the F_{MSY} value of 0.19
- Seal cull + cod recovery: combination of two scenarios above
- Spatial F: stocks distributed on the shelf (cod, haddock and whiting) targeted at F=0.05 (lowest F among these three stocks) while stocks distributed in deeper waters on the shelf-edge (saithe, hake and monkfish) are fished at F=0.19 (lowest F_{MSY} value for monkfish is 0.19). This scenario is a proxy of a spatially based exploitation designed to avoid discards in the mixed demersal fishery.

Table 4.1. FMSY values for west of Scotland stocks

Stock	F_{MSY}	Notes
Cod	**0.19**	
Haddock	**0.37**	Value for areas IV and VI
Whiting	Undefined	Should be as low as possible
Nephrops	**0.116**	Averaged between FU 11 and FU12
Saithe	**0.32**	Value for areas IV and VI
Hake	**0.27**	Value for northern stock
Mackerel	**0.22**	Value for Northeast Atlantic
Horse mackerel	**0.09**	Value for western stock
Blue whiting	**0.3**	Value for Northeast Atlantic
Herring	**0.16**	

Modelling framework

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. EwE is an end-to-end foodweb model representing species as functional groups (biomass pools) and fisheries (for full description see D 4.1). Ecopath is a mass-balanced model of an ecosystem where biomass production of each group balances its losses due to consumption by its predators, fishing, emigration and natural mortality. Fisheries can be represented by one or multiple fleets targeting different groups. See Christensen & Pauly (1992) for a detailed description including equations. Ecosim is a dynamic simulation models which uses parameters and biomasses from Ecopath to simulate the changes in biomasses and fisheries catches over time as a result of fishing mortality or effort. See Walters et al. (1997) for a detailed description including equations.

The EwE model for WoS used in this study was first built by Haggan & Pitcher (2005), then updated by Bailey and al. (2011) and extended by Alexander et al. (2015). The area modelled corresponds to the continental shelf of the ICES area VIa defined by the 200 m depth contour and covers ~110,000 km2 (Fig. 4.1). The model comprises 41 functional groups (see Figure 4.2 for foodweb structure) spanning ~5 trophic levels which include all major commercial fish and shellfish species, their main preys (i.e. small fish and plankton groups) and predators (large fish and mammals), as well as five fishing fleet. Cod, haddock and whiting groups are split between immature (age 0 and 1) and mature (age 2 and above) stanzas. The start year of the model on which Ecopath is based was 1985 while the dynamic component Ecosim was calibrated from 1985 to 2008 (see Alexander et al. (2015) for details).

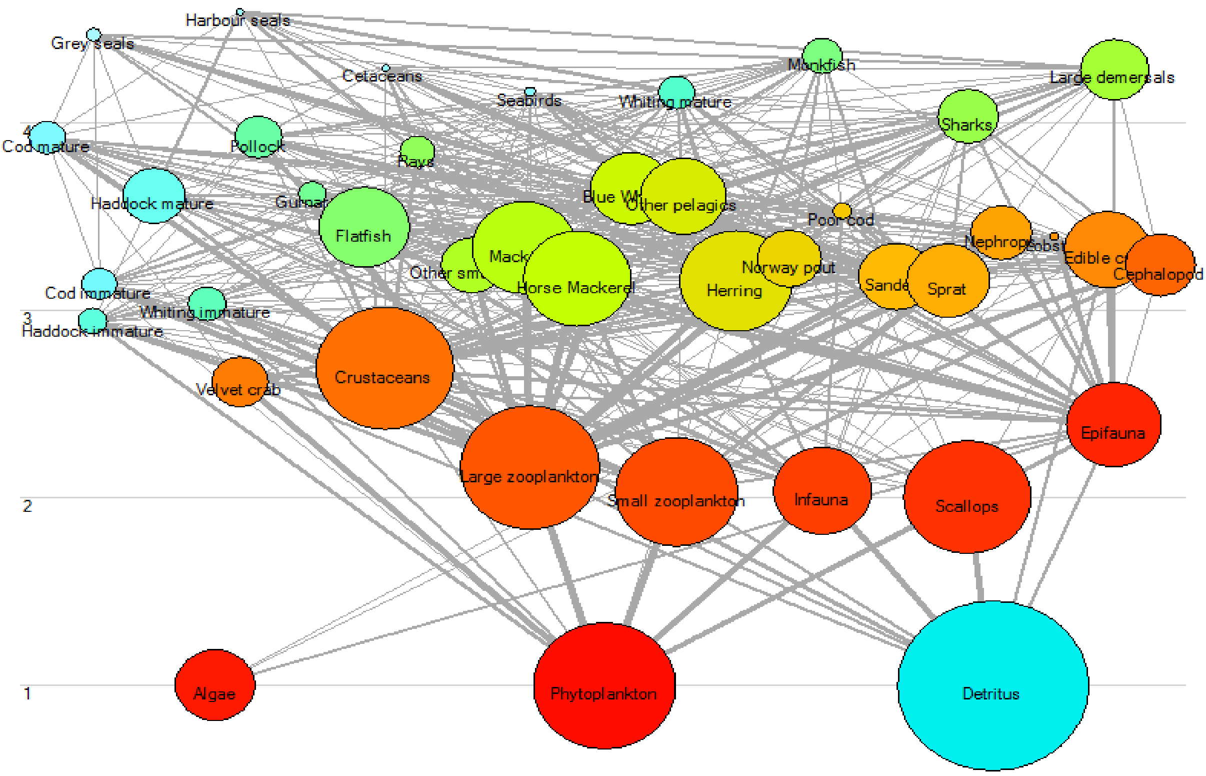

Figure 4.2. Foodweb structure of the EwE model for west of Scotland

The model from Alexander et al. (2015) was revised and updated to extend the parameterisation to 2013. The update of Ecopath consisted in two steps. First, the 1985 biomass starting values of groups for which data was available were updated using the latest stock assessments while the 1985 catches for all groups were updated with the latest landings and discards (when available) data. Secondly, the diet matrix used by Ecopath was updated. Adjusting the diet matrix is a powerful and often underused way of improving EwE models (Ainsworth & Walters, 2015). To improve the goodness of fit, the diet matrix was updated following these consecutive steps: (i) the data and proxies used by Bailey et al. (2011) to build the diet matrix were reviewed (ii) the diet composition of each group was checked individually against existing literature for unusual preys (iii) when unusual prey/predator links were found these were removed and/or amended based on, in that order: available literature; the DAPSTOM database (Pinnegar, 2014); the diet matrices of the EwE models from two neighbouring and closely related ecosystems, North Sea (Mackinson and Daskalov, 2007) and Irish Sea (Lees and Mackinson, 2007).

Ecopath requires numerous parameter values to be entered, many of which are estimated based on best available data when no peer reviewed information is available for the species and area considered (e.g. diet composition). It is therefore up to the user to make sure that the model is ecologically sound. However, as the model is now used widely in marine ecology a pre-balance (PREBAL) has been developed as a standardized method which consists in a series of diagnostics (biomasses across trophic level span 5 – 7 orders of magnitude; the biomass slope on a log sale declines by ~ 5 – 10% with increasing trophic levels; predator/biomass ratios are <1; and vital rates decline with increasing trophic levels) and helps ensuring that the model is ecologically sound (Link, 2010). These PREBAL diagnostics were successfully applied to the West of Scotland Ecopath.

Ecosim on the other hand is fitted using a statistical fitting procedure which minimises the sum of squares between the model estimates and the time series of historical data for both biomass and catches. To update Ecosim, the time series of biomass, catches, and fishing mortalities driving the model were updated (from 1985 onwards) and extended (up to 2013) for as many groups as possible using the latest data available.When fitting Ecosim, catches are considered on absolute scale (i.e. the model aims at replicating the historical catch values) while biomasses are considered on relative scale (i.e. the model aims at replicating the historical trend in biomass rather than replicating the exact values). As a result, biomass time series from survey data were used to fit Ecosim for the West of Scotland in order to capture the trend shown from empirical data rather than from assessment model estimates (the exception to this were cod haddock and whiting for which data was needed for multiple stanzas).

Ecosim includes a 'fit to time series' module which identifies the prey-predator interactions most sensitive to changes in vulnerability (Tomczak et al., 2012). The parameterisation then consists in adjusting these vulnerabilities until the best 'fit' of the model outputs to historical time series is achieved. Goodness-of-fit is assessed by the sum of squared differences between the predicted and observed values on log scale (Christensen et al., 2001). The fitting procedure described in Alexander et al. (2015) was applied. Following the fitting procedure, the best model identified included fishing and trophic effects, but no environmental data (i.e. the inclusion of environmental drivers did not improve the fit). The West of Scotland Ecosim successfully replicates catches and biomass trends (Fig. 4.3 and 4.4).

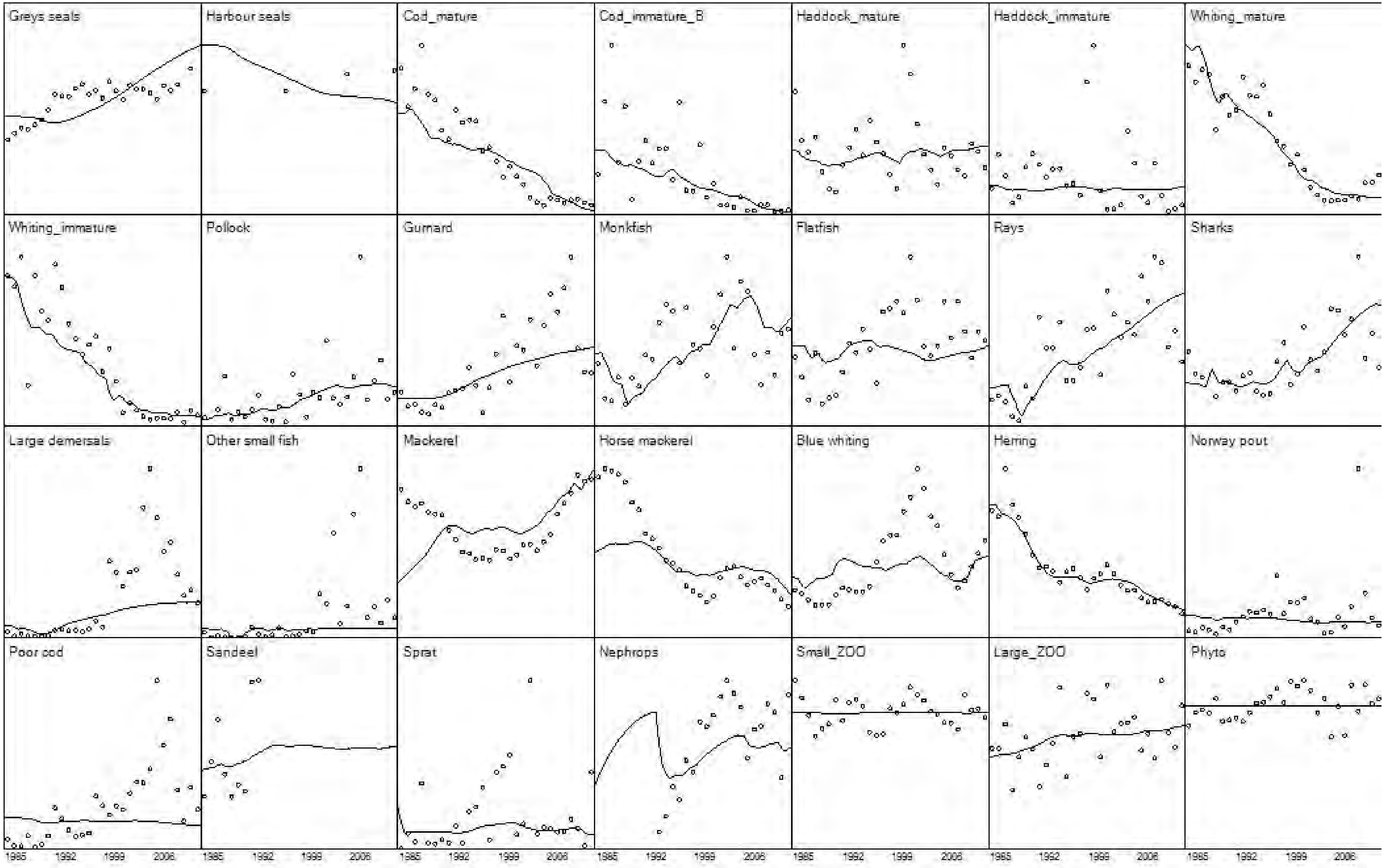

Figure 4.3. Fit of the biomass model outputs (solid lines) to historical data (dots)

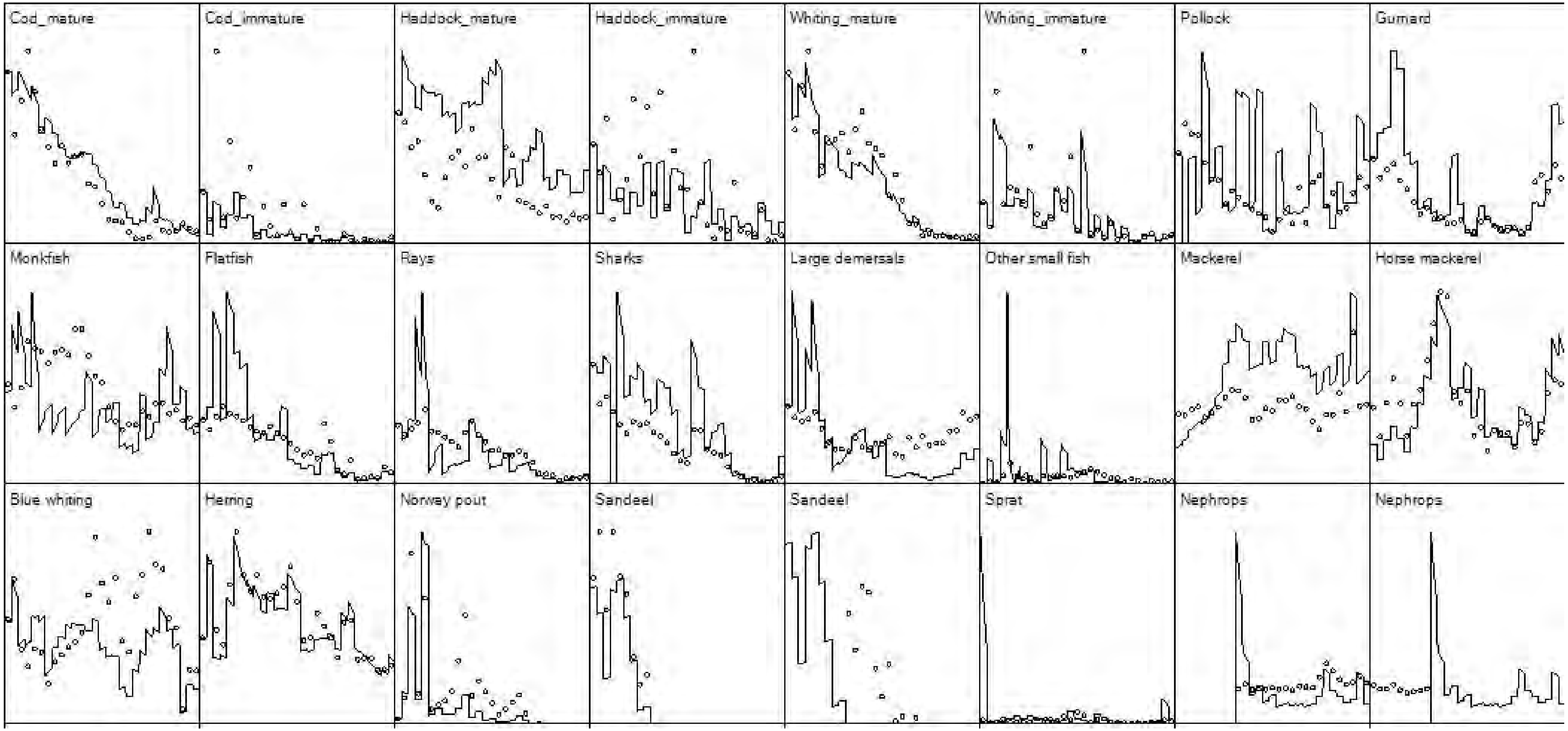

Figure 4.4. Fit of the catches model outputs (solid lines) to historical data (dots)

GES indicators

Contrary to size-based models, EwE does not return size-specific outputs and computing meaningful ecosystem indicators commonly employed such as Large Fish Indicator can be challenging. To investigate whether biomass- and catch-based indicators (Gascuel et al., 2016; Bourdaud et al., 2016; Link, 2005) could be employed to identify the performance of alternative management strategies towards achieving GES, the following ecosystem indicators were calculated using EwE outputs from dynamic simulations to assess GES descriptors 1, 3 and 4, for each simulation performed (see references for details including equations):

- Proportion in weight of large species
- Shannon diversity index
- Mean maximum length
- Mean trophic level (biomass of tropic levels >1 / total biomass)
- Mean trophic index (biomass of tropic levels >3.25 / total biomass)
- High trophic index (biomass of tropic levels >4 / total biomass)
- Apex predator index (biomass of tropic levels >4 / biomass of tropic levels >3.25)
- Pelagic to demersal ratio
- Balance evenness index (Bauer & Bartolino, in prep.)
- Slope of size spectra
- Proportion of flatfish
- Sum biomass of tropic levels > 4
- Sum biomass of piscivorous
- Sum biomass of flatfishes

The reason why so many indicators were computed is twofold: (i) to assess which indicators the model can replicate accurately, (ii) to assess which indicators best captures variations in fishing exploitation patterns.

The performance of the model in replicating these indicators is shown in figure 4.5. While some indicators are replicated fairly well by the model (e.g. MML), other are clearly off-scale (e.g. MTL). However, the model captured the trend for all GES indicators (Fig. 4.5.).

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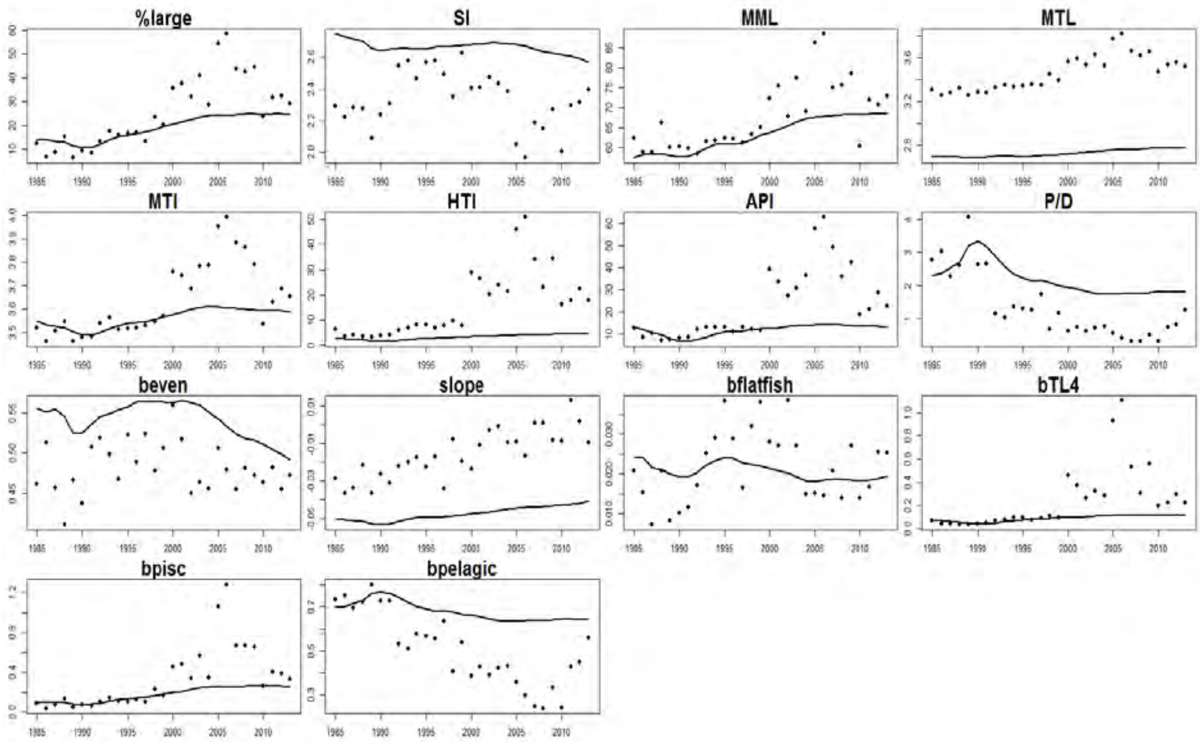

Figure 4.5. Model estimates (solid lines) of GES indicators compared to indicators calculated with historical data (dots).

Socio economic indicators

Economic data was obtained from the Scientific, Technical and Economic Committee for Fisheries (STECF) of the European Commission. Every year, the STECF publishes a report containing a summary of economic performance of EU fishing fleets. The data annexes from this report can be downloaded at https://stecf.jrc.ec.europa.eu/data-reports. The data contained in these annexes include effort, costs, revenue, profit, and landings for each country/fleet/gear combination. STECF data are available from 2008 to 2014, and the monetary unit is €.

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, for each fleet/species combination, using the proportion of landings made in area 6a compared to the landings made in area 27. In addition, it was assumed that: (i) cod, haddock, whiting, saithe, monkfish and hake were caught only by demersal trawl (i.e. TR1), (ii) mackerel, horse mackerel, herring, sprat and blue whiting were caught only by pelagic trawl, and (iii) Nephrops were caught only by Nephrops trawl (i.e. demersal trawl TR2). No distinction is made between TR1 and TR2 metiers of the demersal fleet segment in STECF. As a result, the data extracted for the Nephrops/demersal trawl combination was assumed to be caught by Nephrops trawl only.

Once the data was scaled down, the following was extracted for each fleet/species combination:

- 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

While historical costs are known, future costs over the simulation period are unknown. In order to estimate future costs in fishing scenario simulation, costs coefficients were calculated to relate costs to fishing mortality for demersal species and to landings for pelagic species following the work form Quaas et al. (2012).

For demersal species and Nephrops:

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

For pelagic species:

$$Cost\ coefficient_{species} = {Cost_{pelagic\ trawl,species}}\Big/{Landings_{species}}$$

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

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

And for each pelagic species as follows:

$$Profit_{species} = \left(landings_{species} * price_{species}\right) - (cost\ coefficient_{species} * landings_{species})$$

These profit estimates were then compared to the historical profit time series extracted from STECF data for each species in Figure 4.6.

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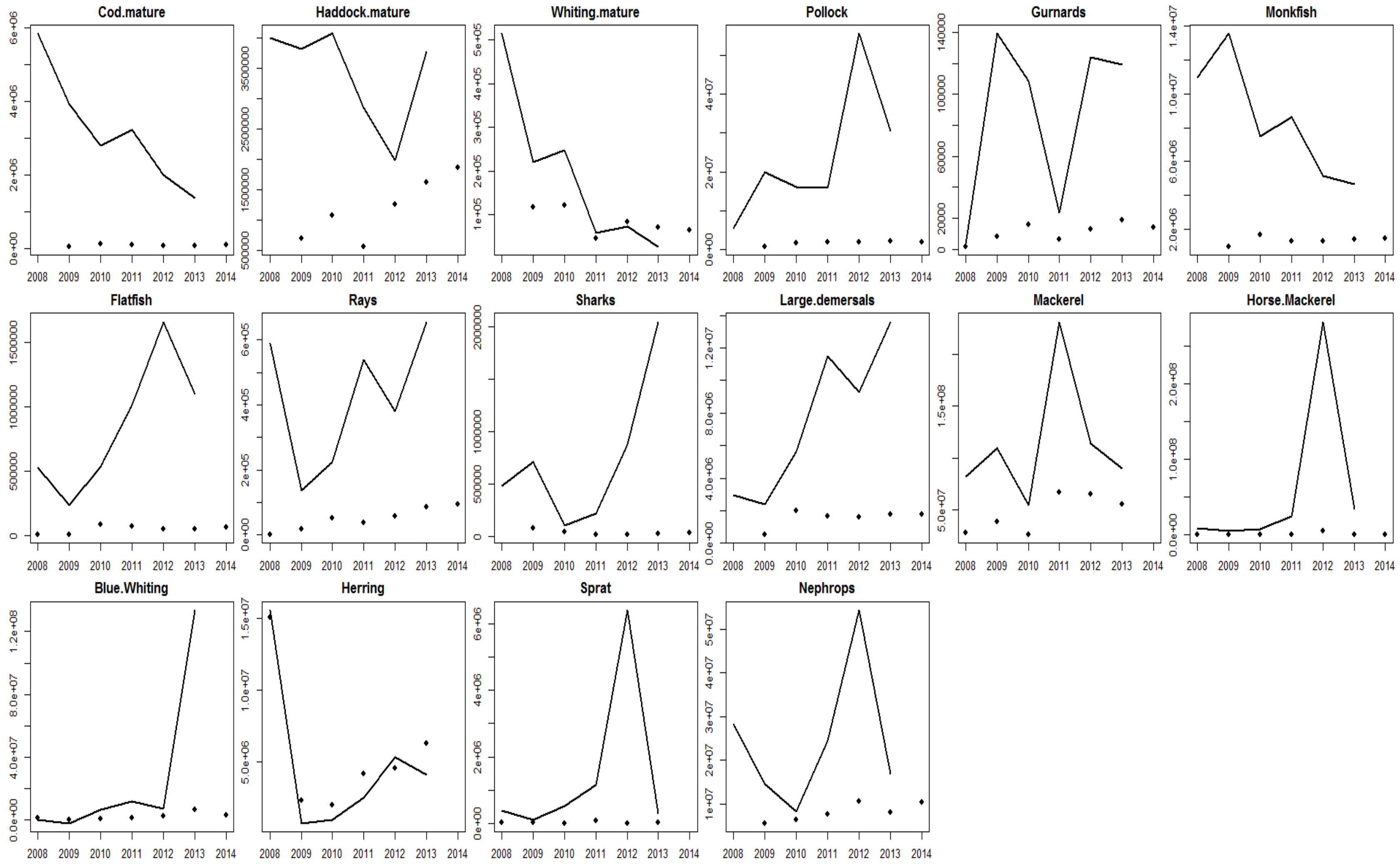

Figure 4.6. Model estimates (solid lines) of profit compared to historical data (dots).

The fit of the profit estimates to the historical data was overall pretty poor, with the exception of herring for which the fit is remarkably good (Fig. 1). In addition, profit estimates seemed to fit somehow better for pelagic species with horse mackerel, blue whiting and sprat showing years for which profit estimates matches the historical values range. For mackerel, although the profit was overestimated compared to historical values the trend did match the data. For demersal species, none of the species showed profit estimates which matched the historical tend and only whiting showed two years with profit estimates matching the historical values range. Lastly, saithe, flatfish, horse mackerel, sprat and Nephrops all showed an outlier in 2012 which seems to indicate an issue with the data affecting both pelagic and demersal species. The fact that the profit estimation seemed better for pelagic species than for demersal ones questions whether using a cost coefficient based on fishing mortality for demersal species, as suggested by Quaas et al. (2012), is in fact appropriate.

A major drawback of the approach followed here is that the resolution of the data available from STECF does not match the resolution of our model area. As a result STECF data was scaled down from FAO area 27 (northeast Atlantic) to the ICES area 6a (west of Scotland). Therefore, the historical data to which the model estimates are compared are proxies themselves. The use of a cost coefficient allows for predicting future costs based on either fishing mortality or landings which in turn allows for estimating profits from fishing scenarios simulations. However, this coefficient relies on one crude assumption: that cost is linearly related to either fishing mortality or landings. The results presented here seem to indicate that such assumption (i.e. cost related to fishing mortality) does not hold for demersal species given the poor fit to historical values observed.

Scenarios evaluation

The seven scenarios mentioned above (i.e. the management alternatives agreed upon with stakeholders, see 'Case study objectives' section) were simulated over a 20 years projection period (2013 to 2033) using the EwE modelling framework described here. For each scenarios, outputs of biomass and catches were plotted over the projection period (see figures 4.7 and 4.8). GES indiactors time series returned by the model were also plotted over the projection period (see figure 4.9). Although the model is not currently able to replicate historical profit time series as discussed above, profit outputs from the model were summed accross fleets as follows and plotted over the projection period (see figure 4.10):

- Demersal trawl profit: sum across demersal species (cod, haddock, whiting, saithe, hake, monkfish)
- Nephrops trawl profit: profit for Nephrops only
- Pelagic trawl profit: sum across pelagic species (mackerel, horse mackerel, herring, sprat, blue whiting).

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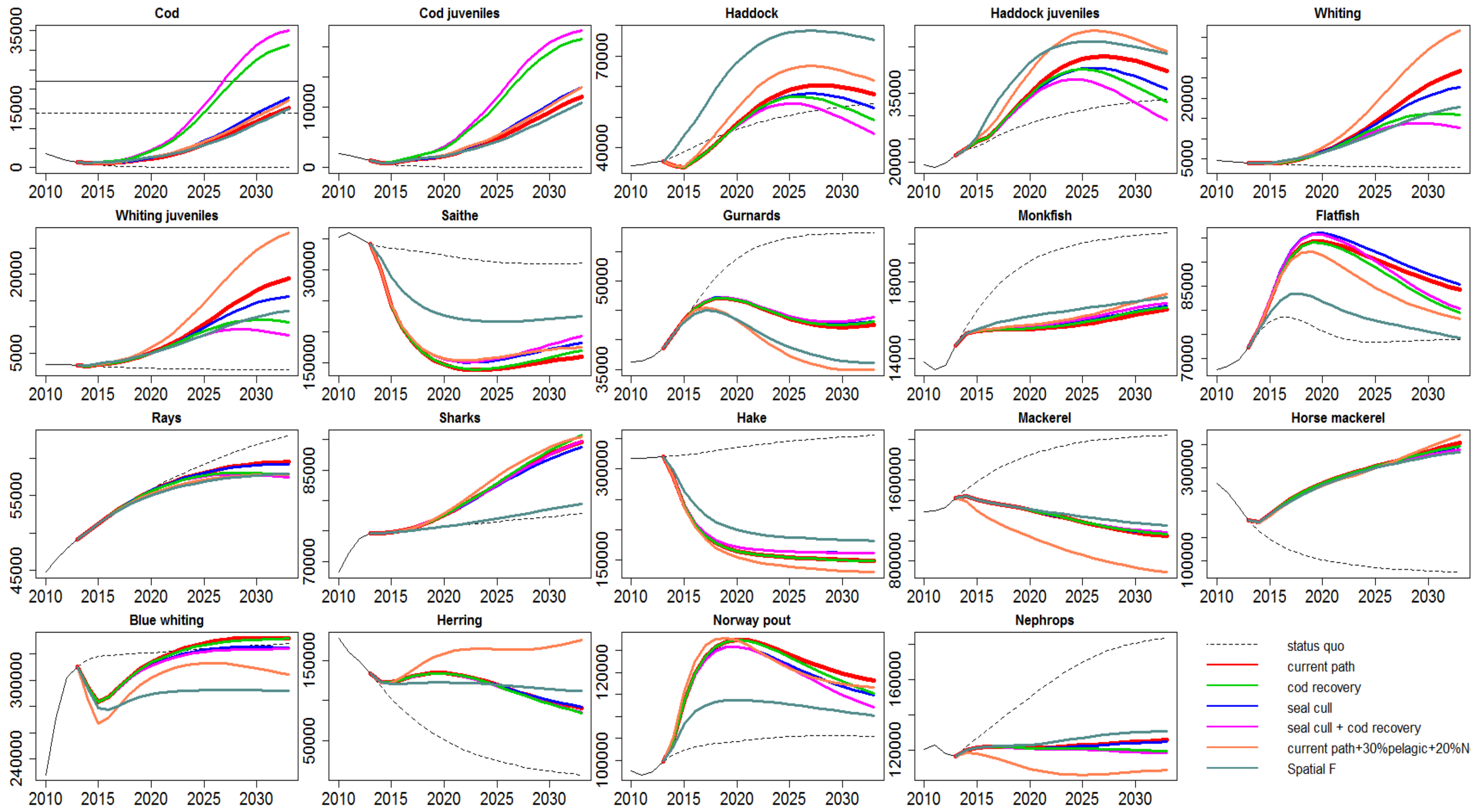

Figure 4.7. Biomass outputs (in tonnes) from the model for the scenarios tested (for cod, the horizontal dashed and solid lines are Blim and Bpa)

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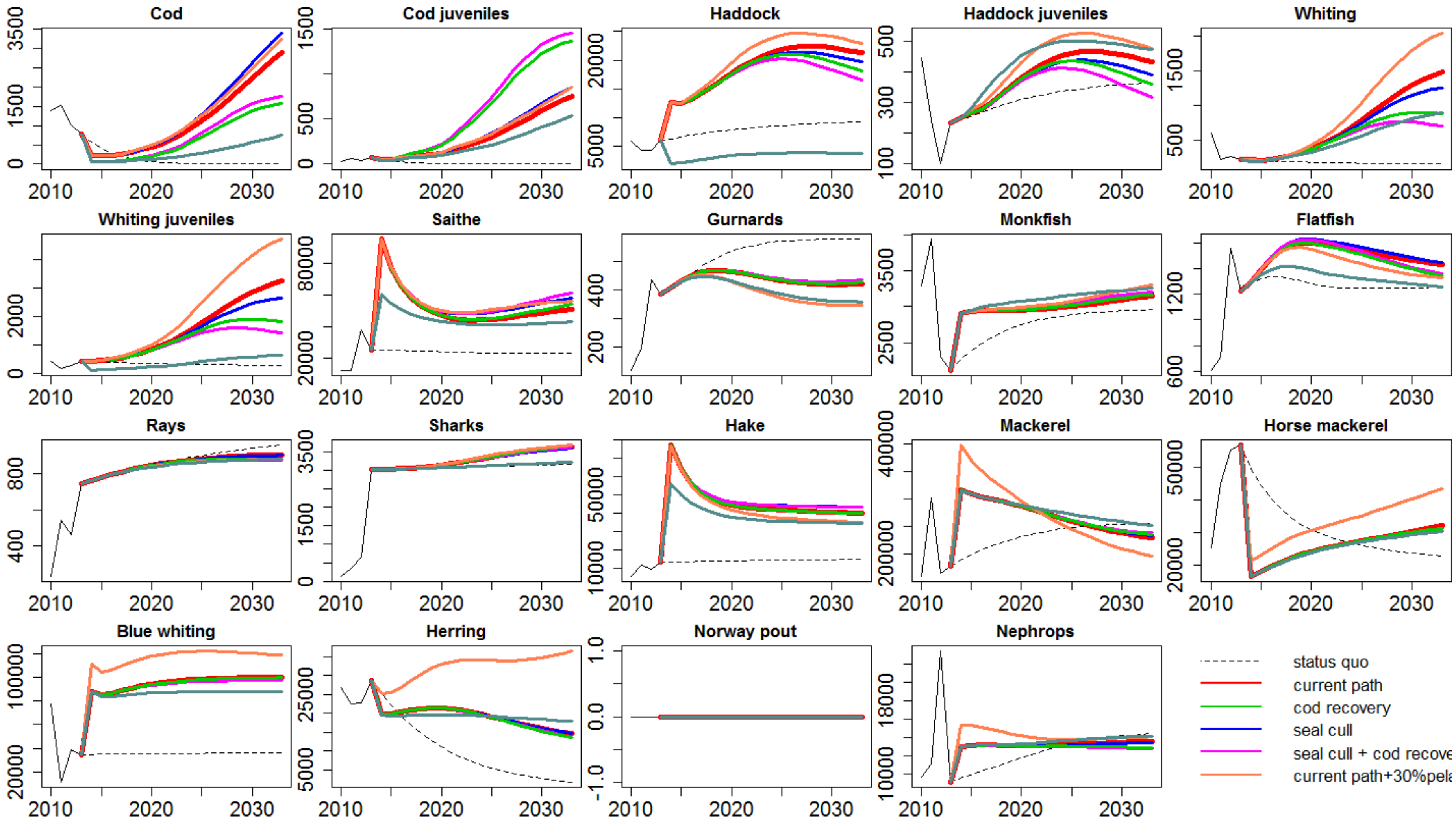

Figure 4.8. Catch outputs (in tonnes) from the model for the scenarios tested

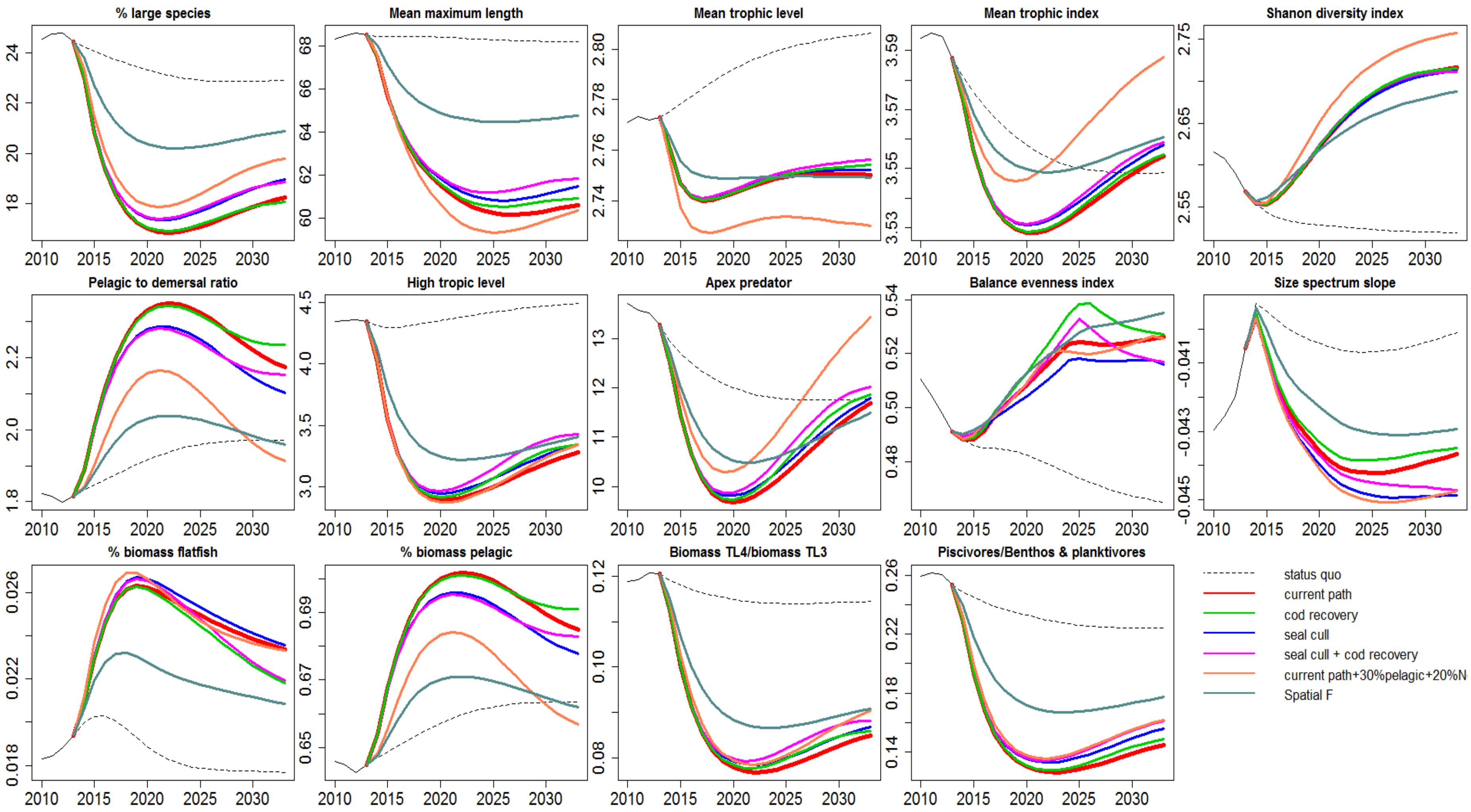

Figure 4.9. GES indicators from the model for the scenarios tested

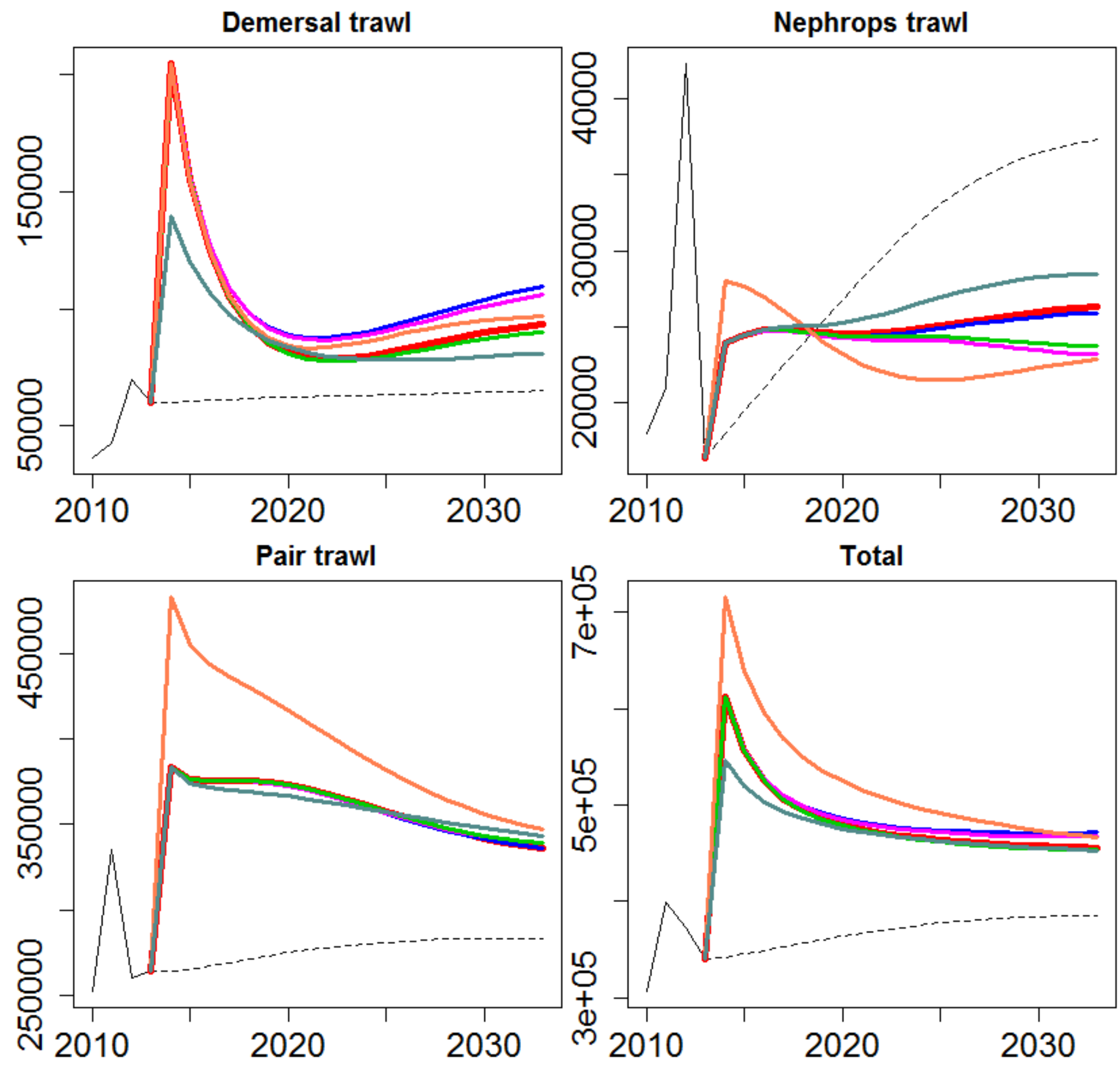

Figure 4.10. Profits (in thousands of €) from the model summed up accross fleets for the scenarios tested

Conclusions

The results presented here highlight the importance of accounting for prey-predator interactions, something which cannot be achieved through single species modelling. In our case, the most obvious example is the effect of saithe predation on juvenile cod which appears to be preventing the cod population from recovering. When fishing at status quo, the biomass of saithe is high while cod juveniles, and subsequently adults, remain depleted (although F status quo for cod is high and therefore probably contributing to the poor stock status). When fishing saithe at F_{MSY} (which is higher than F status quo due to the currently high biomass relative to current F levels for saithe), the saithe biomass is lower while cod juveniles and adults increase in all scenarios. However, in the Spatial F scenario saithe biomass is higher than when fished at F_{MSY} while the cod biomass does not increase as much as the cod recovry scenario despite experiencing the same F of 0.05. This demonstrates the impact of saithe predation on cod juveniles on the cod stock, despite juevnile cod being a small portion of saithe diet. This observation is consistent with the conclusions from the WGECO (ICES, 2015) which identified saithe as an emarging top predator in northen Europe with potential impacts on the foodweb.

The current path scenario appears not sufficient to recover cod, with the biomass of adults (i.e. SSB) just reaching over Blim but still below Bpa at the end of the simulation period. In contrast, applying a F=0.05 (i.e. not targeting the stock) brings the SSB well above Blim. This suggests that in order to recover the cod stock drastic reductions in F are needed. These results are consistent with Alexander et al. (2015). It has to be noted that recovering cod, a top predator in the west of Scotland ecosystem, results in lowere biomasses of haddock, whiting and Nephrops (when harvested at F_{MSY}) due to increased predation. Again, this highlights the importance of considering prey-predator interactions within an ecosystem.

Culling 10% of the seal population every year results in cod recovering slightly faster than without cull with both F_{MSY} and F=0.05. This is consistent with the work from Cook et al. (2015) which concluded that high predation mortality from seals could jeopardise efforts to recover cod. In our case the predation of seals seems to only have little impact. However, the seal diet values in our model are based on 1985 values and somewhat conservative. Diet data from 2002 suggest that seals are currently consuming larger amount of cod, although other fish species are still the bulk of the diet. As a result we are probably underestimating the seal predation on cod.

When considering Good Environmental Status and the overall ecosystem health there does not appear to be a clear winner among the scenarios tested. The spatial F scenario seems to be performing best accrss the board, probably due to the fact that it applies lower F values in general. However, this scenario returns the lowest Shannon diversity index of all scenarios apart from status quo. In terms of usefulness of the indicators tested here, the balance evenness seems to be the most sensitive to changes in fishing exploitation patterns. This is likely due to the fact that this indicator assesses the diversity within each trophic level rather than accross the whole foodweb.

All scenarios tested returned somehow similar profits for the demersal trawl fleet. Unsurprisingly, the spatial F scenario returned the lowest profit since it applies low F values, while the two scenarios where seals are culled returned the highest profit due to reduced predation mortality. For the Nephrops trawl fleet, increasing F by 20% led to an increase in profit in the short term but a decrease in the long term, with the profit on medium and long term being the lowest of all scenarios. In other scenarios it is worth noting that the higher the cod biomass is, the lower the Nephrops trawl fleet profit is, both on short and long term. For pelagic trawls, increasing F by 30% also led to an increase in profit in the short term although this scenario led to a profit in 2033 on par with other scenarios. All other scenarios led to very similar profits for the pelagic trawl fleet. Overall, despite significant differences in fishing mortalities, all scenarios led to similar total profit by the end of the simulation period. As expected, the scenario with an increase in Nephrops and pelagics F led to the highest short term profit but the difference in profit compared to other scenarios becomes negligible by 2033. It is worth mentioning that the spatial F scenario, although applying lower F values, resulted in a total profit on part with other scenarios on the medium and long term. This is likely due to the reduced costs associated with lower Fs. However, the scenario with the highest Fs resulted in the highest sum of profits across the simulation period and as such is most likely to be preferred from a stakehloders perpective.

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5. Iberian waters case study

The South Western Waters (SWW) case study is comprised of two different subcases with two models: (1) in the Gulf of Cadiz, with the aim of modelling the anchovy dynamics, including fishing, and the environmental factors that mainly affect its early life-stages (2) in the whole Atlantic Iberian Peninsula, a model for fisheries-cetaceans interactions where the strategy is modelling the population trajectories of hake including cetaceans as predators. In both sub-cases, the main objective of these models is to evaluate management trade offs and conflicting objectives such as single species, ecological, social and economic targets.

5.1. Bioeconomic model for the anchovy fishery in the Gulf of Cadiz

Case study objectives

Co-creation process has transformed the main focus of the South Western Waters-Iberian Waters-Gulf of Cadiz Case Study (SWW CS) as defined in Mareframe proposal. The main objective now, according to stakeholders needs, is to provide advice for an adaptive management in contrast to the current management based on a fixed quota (Deliverable 6.1). This should integrate the environmental forcing (mainly fresh water discharge from Guadalquivir River, sea surface temperature and intense easterly wind) on the population dynamics and the socio-economic aspects (i.e. income, employment and profitability).

Modelling framework

To achieve case study objectives we have chosen two models, the first one is a kind of minimum realistic model and the second one is a Gadget model. Minimum realistic models (MRM) provide a suitable framework to test the different scenarios previously defined with the stakeholders, with few parameters including the key environmental processes and their effect on a single species. Instead, Gadget allows estimating the current state of the stock with more precision using all the information available regarding age-length-composition of the stock.

The MRM including economic dynamics was chosen as the best one to inform the Decision Support Tool (DST) which is available in Mare Frame Decision Support Framework website: http://mareframe.mapix.com/gulf-of-cadiz-modeloutput.html. All the scenarios were already tested by stakeholders.

Minimum Realistic Model+ economic dynamics

The MRM used is described in Rincon et al 2016. As a result of this MRM, feasible simulated abundance time series under different scenarios can be obtained. This feature has allowed us to modify the operational model in order to get socioeconomic indicators as requested by stakeholders.

Conceptual model

The modified model is the coupling of the environmentally-driven MRM with an economic model to analyze the consequences of different quotas on its biological and economical sustainability. The model provides the framework to simulate different policy options (such as fixed TAC or adaptive TAC) to manage European anchovy in the Gulf of Cádiz, together with an assessment of its performance that combine anthropogenic, environmental and biological factors looking for maximization of economic return and minimization of collapse probability.

The Figure 5.1 shows the schematics of the conceptual model as the coupling of biology and economy submodels. In the biology submodel, the physical environment is considered to influence the process of recruitment during the early life of anchovy while the economic submodel is based on a system where employees and ship owners share a 50% of the Gross Value Added (GVA) net of Social Security contributions. Catches (*C*) and fishing effort (*E*) provide the linkage between biological and economic submodels.

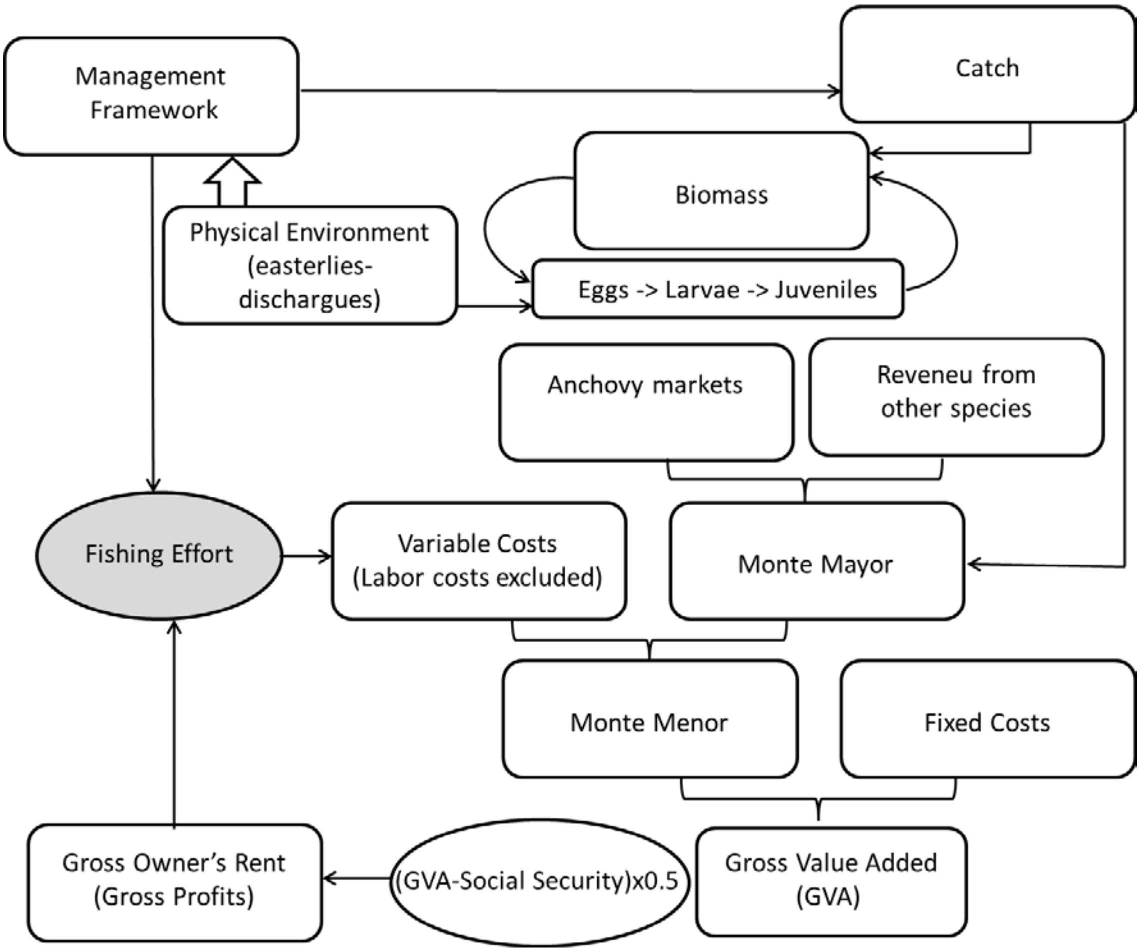

Figure 5. 1: Schematics of the conceptual model as the coupling of biology and economy submodels.

Parameterization, settings and assumptions related to ecosystem processes

The main environmental processes influencing anchovy recruitment as mentioned before are sea surface temperature (SST), fresh water discharge from Guadalquivir River and intense easterly winds.

SST influence is accounted by the number of spawns in a month. Probability of spawning events that occur once, twice, three or four times in a month was calculated from a SST time series available from 1996 to 2004. They were, respectively, 0.37, 0.37, 0.22 and 0.04 during the spawning season from May to September (Rincon et al. 2016). Then to calculate the number of spawns, $s_{y,k,s}$, for year y, month k, and simulation s, we sampled randomly from {1, 2, 3, 4} with the corresponding probability. This parameter is included in computation of the number of eggs as follows:

$$Eggs_{y,k,s} = fec \times sexr \times s_{y,k,s} \sum_{a=11}^{24} N_{a,y,k,s} w_a \quad k=3,\ldots,9$$

Where *fec*= 500 eggs/g, is the number of eggs that a female could spawn per gram, approximated from a review on spawning traits of 22 anchovy stocks in European waters (Somarakis et al., 2004) and consistent with estimates from ICES for the area (ICES, 2006, 2009, 2011 and 2014). sexr=0.5, is the proportion of sexually mature anchovy female anchovy (Millán, 1999) while, w_a and $N_{a,y,k,s}$ correspond to the weight at age and number of individuals, respectively.

The effect of wind and discharges on $\mathrm{Eggs}_{y,k,s}$ is included in the computation of the number of recruits (individuals older than 5 months)

$$N_{6,y,k+5,s} = Eggs_{y,k,s} e^{(-\lambda W_{y,k,s})} e^{(-\lambda W_{y,k+1,s})} e^{(-\lambda W_{y,k+2,s})} \rho_{y,s}$$

Where $W_{y,k,s}$ is the number of days that strong easterlies (> 30 Km h^{-1}) have blown. Each month, $W_{y,k,s}$ is randomly sampled from $N(\overline{W_k}, \sigma_{W_k})$, where the mean and standard deviation are the monthly climatologies of the historical time series. The equation follows the modelling procedure in Ruiz et al. (2009) to represent the negative impact of easterlies during the first three months as a negative exponential with a value of 0.2 for the parameter λ. The time series in Alcala del Río dam (not shown) indicates that roughly one in every 30 years a severe drought force discharges to drop below 10 Hm^3/month during summer. This drop bears a recruitment failure (Ruiz et al., 2006). However, agriculture demands regulate discharges during non-drought years to a rather stable value near 100 Hm3/month thus bringing little variability to recruitment (Rincón et al., 2016). Parameter $\rho_{y,s}$ simulates this based on historical records from 1996 to 2004 by randomly sampling each year between 0 (recruitment failure at drought years) and 1 (no impact on recruitment during normal years) from a probability vector of values 0.03 and 0.97 respectively.

A broad limit of 4.5×10^8 is set to the maximum number of recruits ($N_{6,y,k,s}$) in coherence with maximum abundances diagnosed in Ruiz et al. (2009) and reported by ICES (2014). Similarly, a minimum recruitment of 10^6 is set to reflect the fact that historical records have never evidenced a full extinction of the stock.

From deterministic theory of fishing, adult survival is determined by $e^{-(M+F_{y,k,s})}$ with M = 0.1, and catches are calculated from Baranov catch equation.

The model simulates 1000 iterations of 40 years each. It was initialized with values derived from a run long enough to stabilize the population size. The first ten years of each simulation were discarded to avoid the impact of these initial conditions on the analysis of model outputs.

Scenarios evaluation

Description of scenarios

1. Fixed TAC: Fishing mortality (F) is assumed constant during the year (e.g. 0.5 month-1) but it is reduced when necessary to prevent more landing than allowed by the fixed TAC (6500 tons in year 2015)

2. Adaptive TAC: Fishing mortality (F) is assumed constant during the year (e.g. 0.5 month^{-1}) but it is reduced when necessary to prevent more landing than allowed by the annual adaptive TAC. A default reference TAC* is modified every year using an environmental harvest control rule that takes into account the number of days with easterlies stronger than 30 Km h^{-1} from April to September and the effect of discharges from Guadalquivir river of the previous year according to Figure 5.2. If the previous year discharges are below 10 Hm^3, the reference value, TAC*, is reduced to TAC*/α, otherwise, the TAC* is modified according to the windy days from April to September in the previous year, the TAC* is reduced linearly from α TAC* to TAC*/α, where α is a constant parameter between 1 and 5.

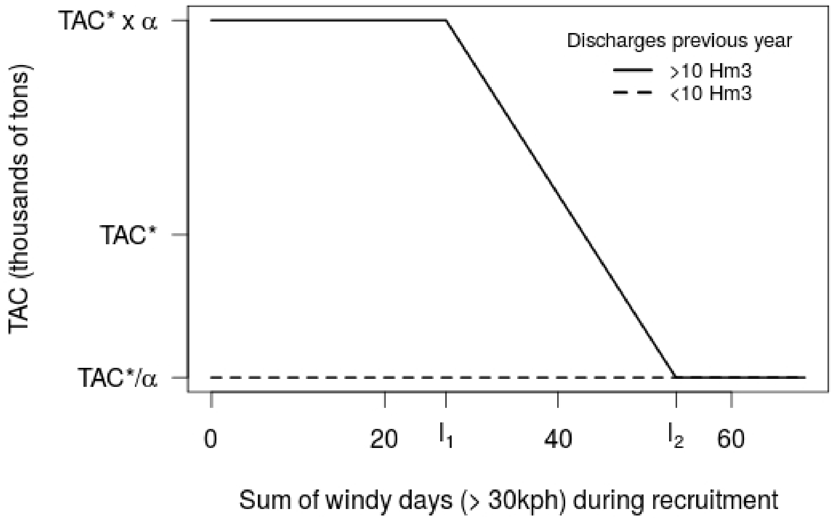

Figure 5.2: Adaptive TAC as a function of the number of windy days during the recruitment period (alpha=1.5, TAC=6 thousand tons).

Ecosystem model outputs

Fixed TAC: Defining the probability of collapse as the proportion of simulations where the average spawning biomass between May and July drops below one thousand tons at least once, this probability abruptly increases at sharp steps with thresholds around TACs of 3000 and 8000 tons as could be seen in Figure 5.3 (triangles). Discrepancies between mean profit (black circles) and assigned TAC become more evident as the fixed TAC increases because the stock is unable to supply the fish that the fleet could nominally catch.

Adaptive TAC: Environmental conditions affecting recruitment (wind and discharges) are known months ahead the fishing season. This knowledge can be used to adjust catches according to the expected evolution of the stock. Figure 5.4 shows the impact of this strategy through the environmental harvest control rule. It represents the average catch and its standard deviation (all years and simulations) versus the parameter α that sets the limits of the fluctuations in the environmental harvest control rule. The different panels indicate different TAC^* values. The average catch is very stable for values of α higher than 1.5 although variability increases with α. The probability of a collapse is not sensitive to α for TAC^* values of 6000 and 7000 tons. However, it abruptly drops to zero when TAC^* is equal to 5000 tons and α is close to 4 or higher.

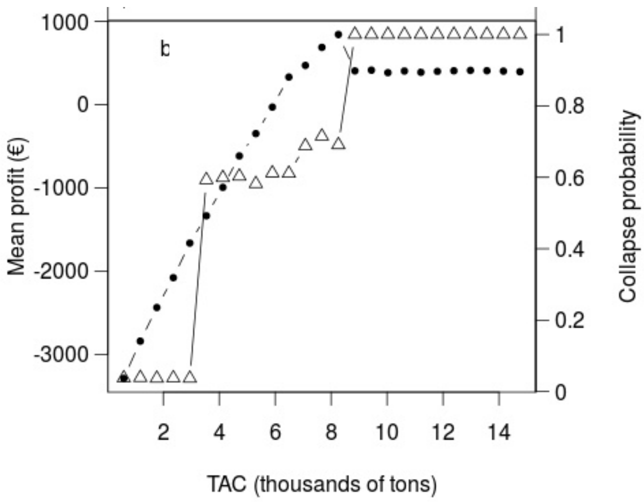

Figure 5.3: Dots and triangles are respectively, the mean (over all years and simulations) of profits for the whole fleet and probability of collapse under different levels of fixed TAC.

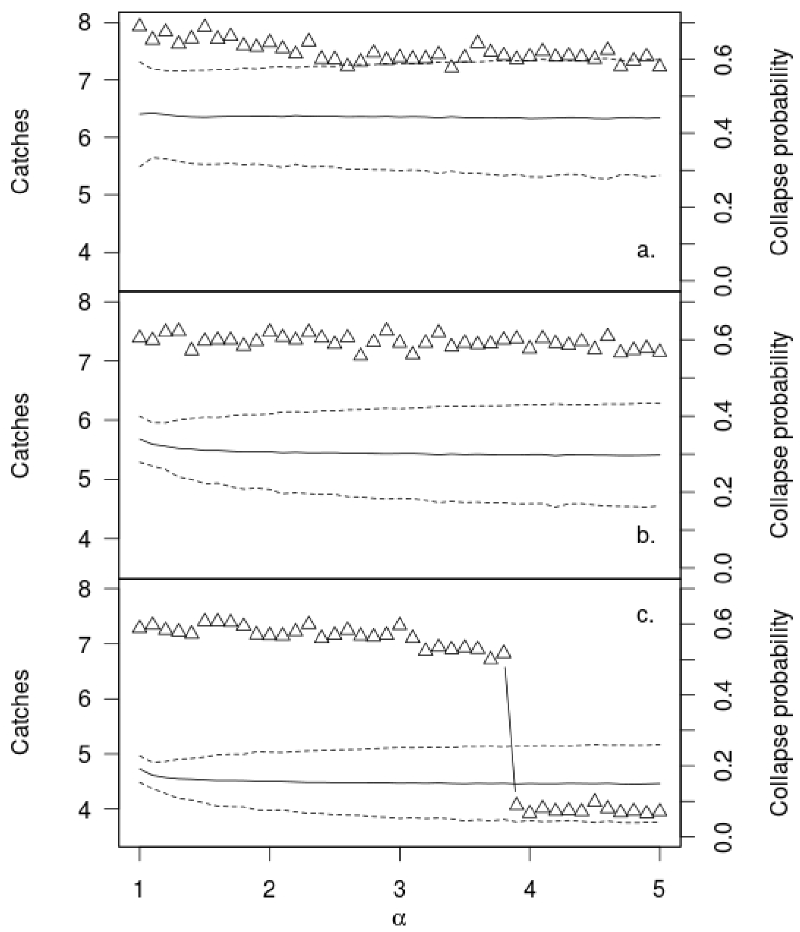

Figure 5.4: Stock dynamics under different values of α for reference value TAC* of 7 (a.), 6 (b.) and 5 (c.) thousands of tons. Solid and dotted lines account for mean, and mean ± standard deviation of the catches (thousands of tons), respectively, while triangles represent probability of collapse.

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5.2. The multispecies model (hake and dolphins) in the Iberian Peninsula.

Case study objectives

In this Case Study we have focused on the southern stock of the European hake (*Merluccius merluccius*) and two cetacean species: common dolphin (*Delphinus delphis*) and bottlenose dolphin (*Tursiops truncatus*). By modelling cetacean abundance, predation and the mortality caused by their interaction with the fishery we can explore the effects of fisheries management measures and the tradeoffs between two different targets, i.e. maximize the fisheries yield and keep dolphin populations in a healthy status.

Modelling framework

SWW Case Study aims to develop a Model of Intermediate Complexity for Ecosystem assessments (MICE) taking in consideration the involved fleets (trawlers, purse seiners, gillnetters, long liners, etc), small cetaceans (common and bottlenose dolphin), the hake and other small pelagic fish. The initial step in the model is the interaction between hake and fleets. At the same time, small cetacean models were developed. Finally, interactions among the different components have been implemented. By modelling cetacean abundance, predation and the mortality caused by their interaction with the fishery we can explore the effects of fisheries management measures.

Gadget

Gadget was the modelling software tool chosen to implement all these ecosystem interactions. It simulates dynamics based on many parameters, and then compares the output data with observed data to get a goodness-of-fit likelihood score. The model runs again with adjusted until it achieves the lowest likelihood score. We chose it because it allows including many species in multiple areas (cetaceans, hake and pelagic fish in ICES divisions VIIIc and IXa), predation between species, maturation, reproduction, recruitment, and, multiple commercial and survey fleets.

Interactions with the stakeholders

Description of the indicators selected to evaluate the good environmental status (GES) of the functional group of marine mammals; belonging to the Descriptor 1 (The biodiversity is maintained). These Indicators include “abundance” and “bycatch rate”. The criteria of GES for these two indicators are that populations should not suffer a significant decline in their long-term abundance, and the by-catch rate must not put at risk the demographic structure of populations, respectively. No reference points or thresholds have been still established for these Indicators criteria. GES of hake descriptors included in the Descriptor 3 (the population of commercial fish species is healthy) are the “spawning stock biomass”, as an indicator of population heath and “fishing mortality” (F at ages 1 to 3), as an indicator of fishing pressure. Reference points considered for this indicator are MSY reference points (F_{msy} and B_{msy}) and precautionary reference points (B_{lim} and F_{lim}). Theses descriptors have already been provided for the period 1982 to 2014 in Deliverable 4.3 and are updated here with a more credible and complete model. Descriptor 4 (Elements of food webs ensure long-term abundance and reproduction) were not considered for this model and the GES of the key predators (dolphins and hake) will be only tested using the Indicators of Descriptors 1 and 3.

Hake is under a recovery plan. Although hake SSB has increased in recent years and is now considered to be at a healthy state, fishing mortality continues being well over F_{msy}. PPC aims to achieve MSY at 2020 as latest. Since 2015 F_{msy} is initially suggested as a target for TAC setting and ICES provides advice based on this approach. For hake this implies a big reduction in TAC. Two meetings with stakeholders and politicians set in 2015 and 2016 suggested the interest in explore the consequences of delaying the hake F_{msy} with different deadlines until the last one, 2020. Hake is one of the most profitable targets in these mixed fisheries, and therefore, the stakeholders’ suggestion is try to keep a most stable yield without compromising the PPC goals, which means to explore different options to delay achieving F_{msy} later, although before 2020.

Conceptual model

The multi-species model developed is a three-species Gadget model that includes the Southern European hake stock, and two species of small cetaceans, common dolphin and bottlenose dolphin, which have been identified to be two important predators of hake (Santos et al., 2014). Moreover, other non-modelled species have been included in the model acting as prey of dolphin stocks, the sardine (Sardina pilchardus) as the main prey of common dolphin and other prey species pooled in a category called otherfood, for both common and bottlenose dolphins. The Southern European hake single-species model employee for the assessment of this stock in 2015 (ICES WGBIE, 2015), benchmarked in 2014 (ICES WKSOUTH, 2014), was used to develop this multi-species model. Efforts of the Spanish and Portuguese fleets included in this model were also linked with the dolphin models to reflect variations in bycatch rates during the time series of data. Consumption of dolphins, amounts and length classes of preys were derived from stomach analysis of stranded animals. Data used for performing these models and its relationships are described in Deliverable 5.2 - Data available for modelling in case study areas.

Alternative models tried

Before linking the three models in a whole multiple-species model, two single-species models were developed for both common and bottlenose dolphins.

The abundance of common and bottlenose dolphins in shelf waters of the Atlantic Iberian Peninsula (less than 500 m of depth) were derived from two abundance surveys SCANS-II (2005) and PELACUS (2007-2014) (Saavedra et al., 2015). However, the variability in the annual abundance did not produce good adjustments in the model because mortalities and recruitments employed did not allow such large changes in the abundance of supposedly closed populations. A deeper study of the strandings data series in the northwest of the Iberian Peninsula was carried out by modelling the monthly numbers of stranded dolphins with meteorological and oceanographic variables, in order to find trends in the abundance of populations (Saavedra et al., 2014a). The results revealed that the referred dolphin populations had remained almost constant over the last 20 years with small cyclical variations. Therefore, it was assumed that the variations observed in the surveys could be due to the possible movements of these animals in and out the study area. For this reason, it was decided to use the average abundance calculated in these campaigns, assuming constant populations over time, which, on the other hand, facilitates the modelling and interpretation of the outputs.

The first option tested for annual population's renewal was to use a constant recruitment, equal to the annual number of deaths produced in both common and bottlenose populations. This scenario, however, did not allow varying the production of stocks according to their abundance. The next step for improving both cetacean models was to apply a fecundity function for female stocks. The only dolphin stock was divided in immature dolphins, males and females, moving individuals from the first to the seconds with a sex-ratio of 1:1 when achieving the maturity, estimated from a maturation ogive fitted to the data resulted from the analysis of female ovaries (see Deliverable 5.2). In addition, this made that the three split stocks were modelled with different values of growth and length-weight relations, which made the models more precise. The proportion of females that get birth annually was also estimated from own samples and contrasted with the bibliography in order to apply density-dependent values that affect the fertility of the population within reasonable limits.

Total mortality-at-age for each species of dolphins were derived from population structures observed in a long time series of strandings of the northwest of the Iberian Peninsula described in Deliverable 5.2. Furthermore, these total mortalities were later divided in natural and bycatch mortalities by modifying a standard Heligman-Pollard model to be used with strandings data (Saavedra et al., 2014b, 2014c). This allowed applying a constant natural mortality-at-age for each cetacean species and let the model estimates the proportion of bycatch necessary to keep the population stable, attending to the maturity and fecundity values employee as well as the density-dependent phenomena that slightly modify the stocks fecundity.

All the alternative dolphin models were liked with the single hake model by using the energetic requirements of both common and bottlenose dolphins related with their individual weight and the proportion of preys found in the stomach contents of the analysed stranded dolphins. Preferences for each prey species were iterative set until the desired proportions of consumption were achieved. Preys employed in the common dolphin model were the European hake (modelled stock), sardine and otherfood (non-modelled stocks). Prey species included in the bottlenose dolphin stock were European hake (modelled stock) and otherfood (non-modelled stock). Both so-called otherfood stocks have been used as an artificial set of all other species of prey of which dolphins feed on, whose abundance is constant throughout the study period and their purpose is to provide plenty of food for dolphins whose consumption varies depending on the relative abundance of the other prey populations.

Model selection

The Gadget model is quite unstable when is allowed to estimate the natural mortality of the stock. The natural mortality rate used for the assessment of the European Hake stock is 0.4 for every age class (see ICES WKSOUTH, 2014). Since the previous natural mortality (M) represented all sources of natural mortality including predation, it had to be reduced to a given but unknown mortality (M1) once its main sources of predation (M2) were included in the model. A set of multi-species models, with natural mortalities (M1) for hake ranging from 0.05 to 0.4 with an increment of 0.05 each, were tested. A combination of different criteria was used to select the most appropriate M1. First, an estimation of the likelihood was presented for the whole model fit for each M1 figure (Table 5. 1). However the best figures were found at extreme values of M1 without a clear minimum inside the credible range. Second, a comparative approach that relates the total accumulated M at age with the hake assumed longevity. Following Hewitt and Hoenig (2005) define longevity as an age at which the reduction in abundance in 1.5% of abundance at age 0. Then, the longevity corresponding to a constant M-at-age equal to 0.4 year -1 is 11.5 years, which is agreement with the hake longevity assumed under the fast growth model based on tagging experiences (see hake annex in ICES, 2015). The multispecies model provides a different share of M-at-age since cetaceans feed on small hake (mainly smaller than 30 cm, i.e. ages 0 to 3). Assuming than hake longevity do not change in the multispecies model, the most adequate M1 would be this that provides a total accumulated M-at age (M1+M2) for ages 0 to 11 equal than the M assumed. Results and best model selected are showed in Table 5.1.

Table 5.1: Likelihoods and mean natural mortality for ages from 0 to 11 for the set of models tested with M1 ranging from 0.05 to 0.4 with an increment of 0.05 each. The model selected is highlighted in blue.

M1	0.05	0.10	0.15	0.20	0.25	0.30	0.35	0.40
Likelihood	1105	1020	1052	1001	971	970	980	960
B(M1+M2)	0.40	0.39	0.43	0.45	0.49	0.52	0.56	0.60

The best model selected attending the criteria described above was the one with the M1 equal to 0.20. The reason was that the 0.20 model was the one with the most reasonable value of M among those with a smaller likelihood than the previous single-species model (likelihood = 1015).

Settings and assumptions

The main assumption of the model selected is that populations are closed populations. The model assumes that there is no migratory flow between the southern and northern stocks of hake, or between dolphin stocks with the rest of their populations. Moreover, the abundance of dolphins stocks has been stabilized over time and only varies when the model is projected under different scenarios. The abundance used was calculated only in shelf waters, assuming that dolphins inhabiting this area have complete access to the prey species distributed in this area, not deeper than 500 m. The natural mortality of hake (excluding predation) has been considered constant for every age class as in the original single-species model. Natural mortality of dolphins is also constant for the study period but different for every age class. However, bycatch mortality was estimated as equal proportion for every age class and sex.

Hake model is fully described in the hake stock annex in (ICES WGBIE, 2015), the only difference regarding this model is the natural mortality, than now is separated in two parts: a constant M-at-age (M1=0.20) and a variable M-at-age (M2) for small ages that depends on the cetacean population size and available food of others preys. This M2 mainly affects ages 0 to 3.

For both dolphin models all parameters, except bycatch rate, were fitted using own data or from the bibliography. The preferences for the different prey species were iterative estimated until achieve similar proportions in the diet along the whole period as the ones derived from the stomach content analysis. Growth of both cetaceans were fitted to our own data using a Von Bertalanffy model, the weight-length relationships also fitted to an exponential model, mortality at age derived from the strandings population structure. Both maturation ogives were also performed using own data and fecundity estimated. However, this parameter was fitted using own values contrasted with the bibliography and a density-dependent functionality was applied to allow slight variations of this value.

Parameterization and model fitting

Although several parameters of cetaceans models were estimated in different tries, the only parameter estimated in the final model of both common and bottlenose dolphins was the bycatch rate, since models were not robust enough to multiple parameters estimation due to lack of an adequate amount of input data. The bycatch rate was forced to maintain the population abundance stable over time by reducing the likelihood of the surveys abundances estimates, which were established as the mean abundance estimated for every year of the data series. Bycatch rates were first estimated in the cetacean's single-species models and later fitted in the multi-species model. The parameters estimated in the multi-species models were all parameters described for the single-species hake model in the hake assessment and benchmark documents (ICES WGHMM, 2012; ICES WKSOUTH, 2014) excluding the linf for the hake growth model and the natural mortality.

The overall likelihood in the adjustment of the single-species hake model in the 2015 assessment was 1015, while the likelihood as a result of including two species of cetaceans in the same model was 1001, having estimated the same parameters and using the same input values and likelihood components. This suggests that the fit of the model has improved by applying a different mortality for each age class of hake instead of a constant mortality for all of them, as a result of the depredation of the dolphins over the young age classes. Summary plots of the selected model are shown in Fig. 5. 5.

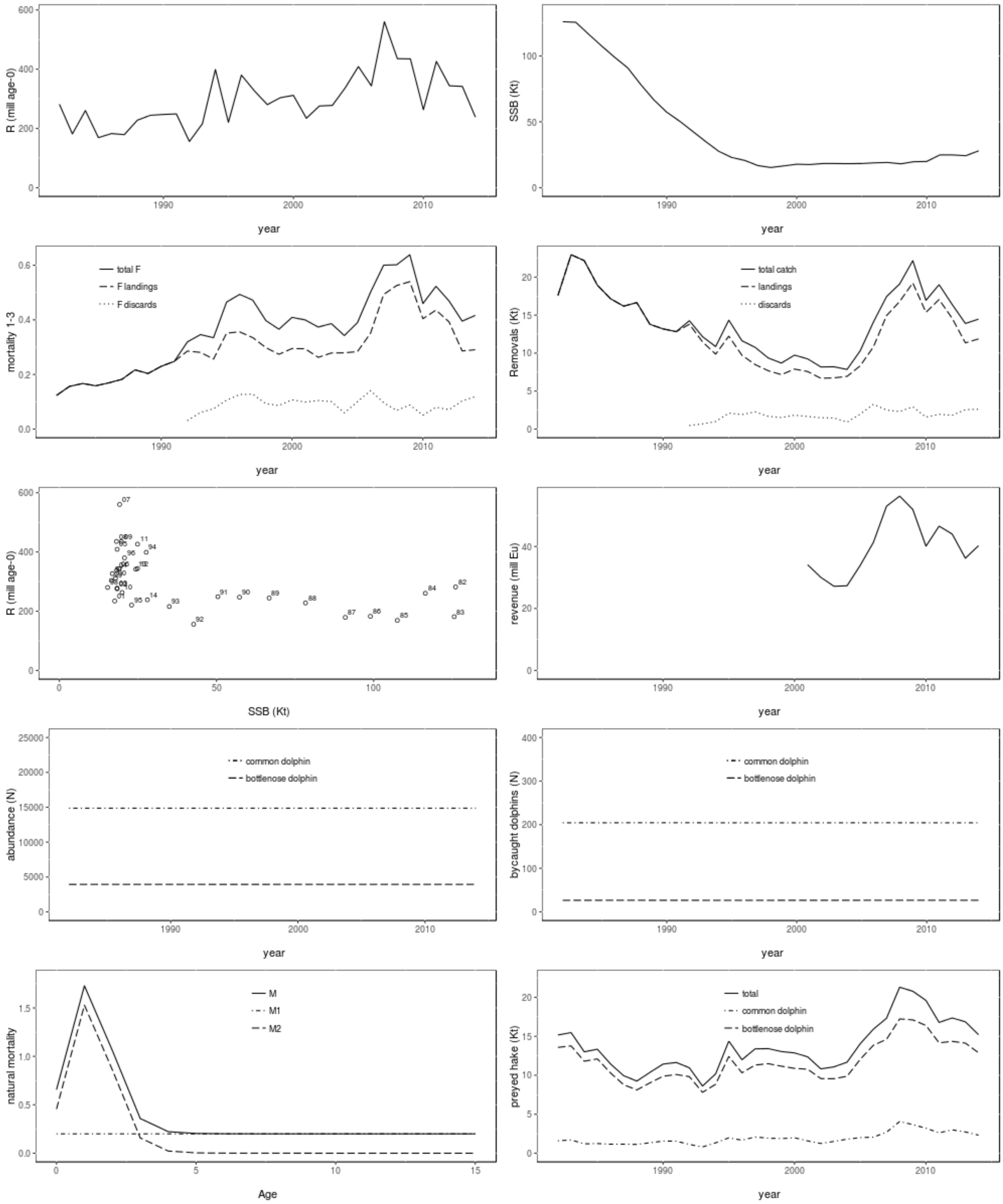

Figure 5.5: From top left to down right, description of summary plots. Time series of recruitments. Time series of SSB. Total, landings and discards fishing mortality. Total, landings and discards removals by fleets. Stock recruitment relationship. Revenue evolution. Common and bottlenose stable abundance. Common and bottlenose stable bycatch amounts. Natural hake mortality (M1 and M2). Total, commons and bottlenose dolphins predation on hake.

Scenarios evaluation

Hake reference points

Populations were projected until 2100 and hake F_{max} (0.15) was estimated as a proxy for F_{msy} (Figure 5.6). Gadget is a length-age based forward projection model, structured by quarters for the southern hake stock. Two different fleets have been used for projections, landing fleet with a logistic selection pattern, and discards fleet with an Andersen selection pattern. Other two additional fleets for common and bottlenose dolphins were also forwarded. A set of multipliers were applied to the current fleet effort to which the stock of hake is subject, ranging from 0 to 2 each 0.01 (for deeper explanation, asumptions and considerations see ICES WGHMM, 2012).

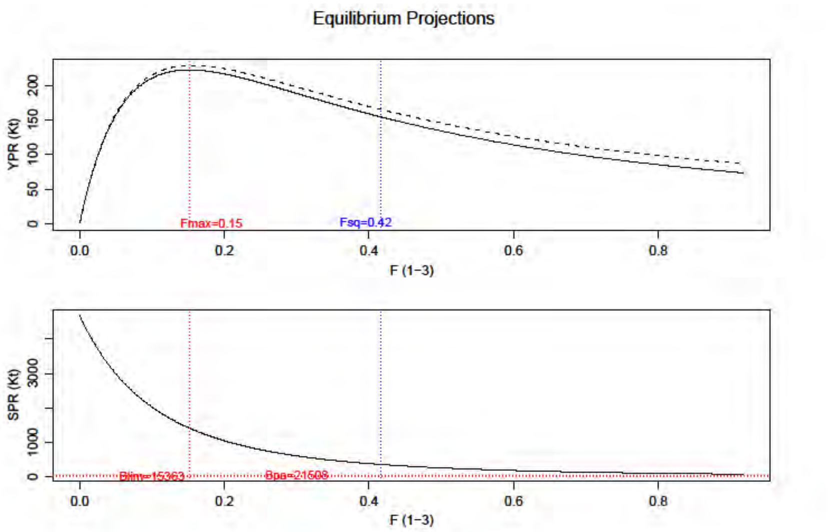

Figure 5.6: Long term projections of YPR and SPR (Kt) with Fmax, Blim and Bpa estimates (t).

Stock recruitment Blim (Bloss = min SSB) and Bpa (Bloss * 1.4) were also estimated following the new hake population parameters. Previous Blim and Bpa were 8000 and 11000 respectively (see ICES WKSOUTH, 2014) and with the multi-species model the new estimates are Blim = 15363 and Bpa = 21506 tonnes (Figure 5.5).

Projection scenarios

Medium term projections with different scenarios of F_{sq} and F reduction were evaluated to achieve the hake F_{msy} in 2016, 2017, 2018, 2019 and 2020. An aditional scenario was added keeping F_{sq} along the projections. In order to define the new F trajectories a constant linear reduction with equal steps each year was proposed, starting in 2015 with F_{sq} (0.42) and finishing in F_{msy} (0.15) each year (2016-2020). The equation to calculate the F in each scenario and each year is the following:

$$F_t = F_{t-1} - [(F_{sq} - F_{msy})/n]$$

Where F_t is the F each year, F_{t-1}(F in previous year); F_{sq} = 0.42; F_{msy}=0.15; n: is the number of years to achieve F_{msy}.

Forward projections settings and assumptions

A similar approach than those used by (ICES WGBIE, 2015) was applied. Recruitment equal to geometric mean of years 1989-2014; maturity at length equal than the mean of last 3 years (2012-14); selection at length equal to the mean of last 3 years (2012-14); ratio landings discards equal to the mean of 2012-14; fishing mortality of the intermediate year (2015) equal to the mean of last 3 years.

For projecting populations under different effort scenarios, the effort to which the stock of hake is subjected was related to the bycatch that produces stable populations of dolphins. In this way, the different scenarios of variation of effort to which the stock of hake was subjected in the short, medium and long term also affected in the same way to the abundance of cetaceans, with consequent variation in the consumption of hake that this might cause in the stock.

Scenarios results

The different medium term scenarios and parameters mentioned above are showed and described in Figure 5.7. Trend of Spawning Stock Biomass (SSB), fishing mortalities (F), catches of the fleet, cetacean abundances, and economic revenues have been evaluated under the various scenarios.

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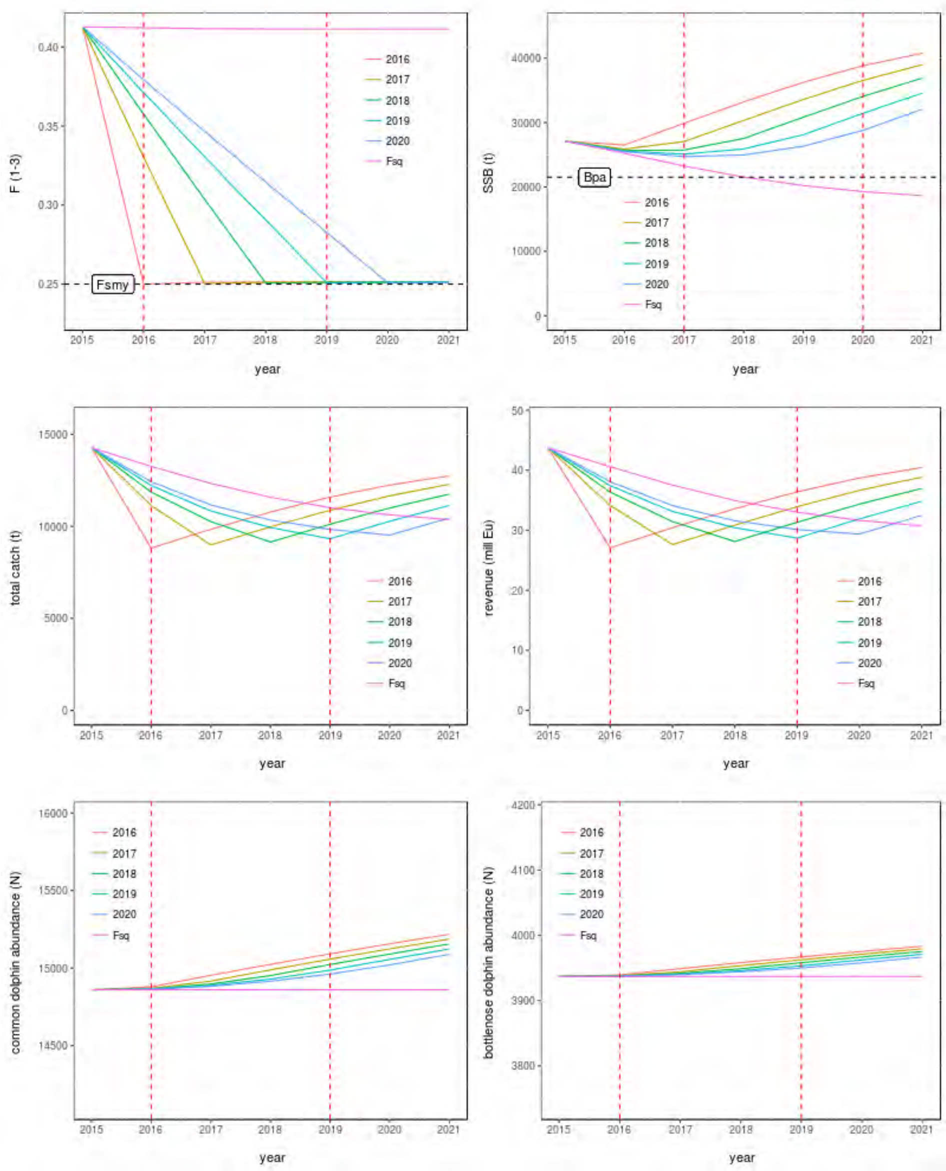

Figure 5.7: From top left to down right, description of plots with Fsq and F reduction scenarios to achieve the hake Fmsy in the various years. F reductions. Spawning stock biomass (SSB) trends. Total catch trends. Revenue evolution. Common dolphin abundance trends. Bottlenose dolphin abundance trends.

The resulting F_s in each scenario can be seen in Figure 5.7. The resulting F for the year TAC (year 2016) are highlighted with a vertical line. Obviously, the more is delayed to achieve F_{msy} the lower the F in 2016 (Figure 5.7 upper left panel). The SSB trajectory when keeping F_{sq} is continuously decreasing, reaching figures below Bpa in 2018 (Figure 5.7 upper right panel). Whe, F is reduced the SSB decreases slightly in 2016 but increases afterwards. The sooner you start decreasing F, the farter the recovery. Catch and revenue are presented in the middle panels. Both present a similar behaviour, i.e. the stronger the F reduction in 2015, the higher the catch and revenue in 2016. However the 2020 expectation is the opposite. Cetacean abundance trends are presented in the lower panels. Bothe dolphins abundance keep stable under Fsq and increased up to a under F reductions with higher increase when the F is lower.

From scenarios outputs we can conclude that:

- The delay in the implementation of F_{msy} implies a lower reduction of F and a higher TAC in 2016. In this way expected catches and revenues are more stable.
- However the SSB recovery and the expected catches and revenue at medium term (2019-2020) are lower, This is the cost of a F_{msy} delay politic.
- With the current cetacean model, with a stable trend where production and by-catch are balanced, keeping F_{sq} would keep their abundance stable. Any other scenario with a F reduction will project their abundance increasing.
- Next step is to compare these projections with those performed when only hake is considered (with M=0.4) to better understand the impact of the fishery in hake advice when cetaceans, as hake predators are considered.

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6. Strait of Sicily case study

Brief description of the case study objectives

The Strait of Sicily (SoS) case study (CS) focuses on the development of a reliable tool for the implementation of the ecosystem approach to fisheries management (EAFM) in a key fishing area in the Mediterranean Sea. The objectives of the CS have been progressively refined through the application of a co-creation approach with key stakeholders (i.e. fishers and fishers representatives, managers of local and national administrations, conservation NGOs, FAO and GFCM officers) and taking into account the objectives of the GFCM international management plan for bottom trawl fisheries exploiting deep water rose shrimp (DPS: *Parapenaeus longirostris*) and hake (HKE: *Merluccius merluccius*) in GSAs 12-16 (GFCM, 2016[4]). Trawl fisheries are the most important in the SoS region both from a socio-economic point of view and considering their impacts on the ecosystem. Trawlers from different nations (i.e. Italy, Tunisia, Malta, Libya, Egypt) and under different management regimes exploit shared stocks in national and international waters thus making challenging the implementation of agreed management rules.

In particular the GFCM plan includes the establishment of three FRAs (Fisheries Restricted Areas) and the closure of the Gulf of Gabes (GSA 13) for three months in summer with the objective to achieve F_{msy} for HKE and DPS by 2020.

The overall goal of the CS has been adapted to provide a tool for the application of EAFM in the SoS which can support the achievement of long term sustainability finding a balance between ecological and human well-being through good governance. In turn the CS also might also substantially contribute to the development of the GFCM management plan through the inclusion of a more holistic approach. ATLANTIS (Fulton et al., 2004) and GADGET (Beagley and Howell, 2004) ecosystem models have been developed to provide advice on the effects of different management scenarios in relation to the following four main management objectives identified 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 (Table 1).

An operating environment supporting continuity of the fisheries is fundamental and it contains economic, social, and resource considerations. Management strategies will respect F_{MSY} targets and, at the same time, take into account the impact of the applied management measures on other ecosystem components and possibly the effects of the ongoing climate change. To this aim, ATLANTIS and GADGET have been be implemented to investigate the direct and indirect effects of multi-fleet and multispecies fisheries on the ecosystem and food web functioning of the Strait of Sicily. Models

4

GFCM, 2016. REC.CM-GFCM/40/2016/4 establishing a multiannual management plan for the fisheries exploiting European hake and deep-water rose shrimp in the Strait of Sicily (GSA 12 to 16) focus on trophic flows between components of the ecosystem, in particular hake, horse mackerel (HOM: *Trachurus trachurus*) and deep water rose shrimp to improve the understanding of the dynamics of these stocks under different scenarios. The impact of alternative technical measures (e.g. area closure, mesh sizes, gear restrictions), on the ecosystem and fisheries is explored in connection with WP6.

Modelling framework

Both ATLANTIS and GADGET have been identified as suitable models to support the implementation of EAFM in the SoS region. The first is a complex ecosystem model able to represent the high complexity of the Mediterranean ecosystem and forecast the impact of management measures on key ecosystem processes, functional groups, populations and fisheries (Fulton et.al, 2004). GADGET (Beagley and Howell, 2004) is a parametric forward-simulation model of an ecosystem, typically consisting of various fish populations, fleets and their interactions. Plagányi (2007) has classified GADGET as a "Minimum Realistic Model (MRM)" to describe the concept of restricting a model to those species most likely to have important interactions with the species of interest. The author stated that "GADGET is currently the model with the most rigorous statistical framework for developing multi-species based management advice. It is also the modelling approach most capable of detailed sensitivity investigations to alternative growth, consumption and recruitment formulations".

On the other ATLANTIS allows to explore the effects on target stocks and the other functional groups included in the model, considering key ecosystem processes (i.e. production, consumption, etc,) and providing a more holistic view on the impacts of simulated management measures. In addition, ATLANTIS is explicitly spatially structured thus allowing to explore the effect of spatially-based management measures. As pointed out by Link et al. (2010), ATLANTIS as a strategic tool is not appropriate to be used in direct support of tactical management decisions (i.e. catch quota, TAC, etc.). Rather, ATLANTIS models can have a relevant role for strategic advice for ecosystem based management due to their ability to synthesize across a wide range of information and simultaneously explore the relative prominence of processes suspected to influence marine ecosystem dynamics (Link et al., 2010).

GADGET and ATLANTIS have been customized to provide management advice in the region on the effects of alternative management measures identified during 3 CS meetings organized in close collaboration with WP1 and WP 6 (Table 6.2.1). In addition, the objectives and results of SoS CS were presented also at international level during FAO and GFCM working groups (Table 6.2.1). Interaction with stakeholders has allowed to set up the models to address the effects of different scenarios of technical measures in the region, including the FRAs (Fisheries Restricted Areas) identified by the GFCM management plan, to achieve the identified strategical objectives. A schematic representation of the general modelling approach followed to develop a toolbox to support the implementation of EAFM in SoS area is showed in Fig. 6.2.1.

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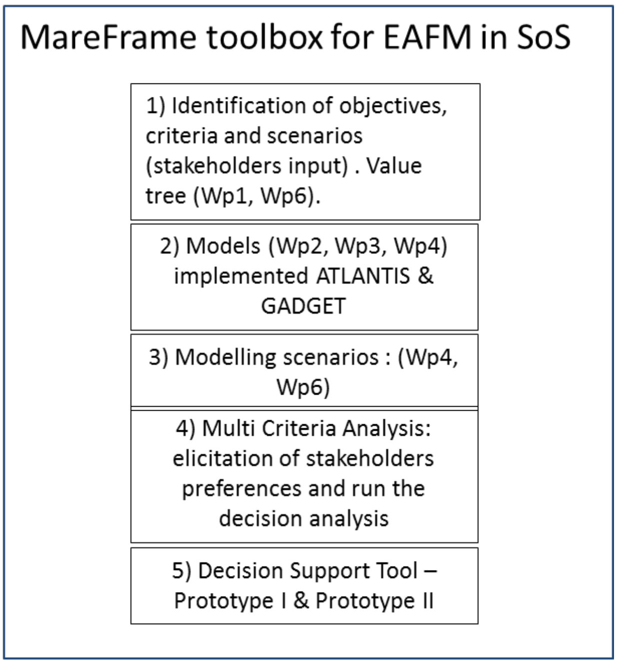

Fig. 6.2.1. MareFrame toolbox for the implementation of EAFM in SoS case study

Table 6.2.1. List of the meetings where the SoS CS was presented and discussed

Type	Date	Venue	Participants	Objectives	Results
CS meeting	20/05/2014	Mazara del Vallo (Italy)	n. 25. Fishers representatives, Vessels owners, NGO (Greenpeace), FAO MedSudMed officer, MEDAC, Sicily Region officers, CNR staff	To identify the main issues of fisheries in the area.	The loss of productivity of the Sicilian fishing enterprises, mostly due to a combination of overfishing, increasing costs and poor market conditions identified as the main issue.
CS meeting	20/02/2015	Mazara del Vallo (Italy)	n. 31. Fishers representatives from Italy and Malta, Fishers, NGO (Oceana), FAO MedSudMed officer, MEDAC, Sicily Region officers, CNR staff, MareFrame researchers, Tunisian researcher	To define the main goals and objectives of fisheries manageme nt in the area.	Four main management objectives identified. Suitable management measures (scenarios) discussed and defined (i.e. effort reduction, fisheries restricted areas, improved gear selectivity, same rules for all, etc).
FAO MedSud Med project meeting	17-18/03/2015	Tunis	n. 35 participants. Fishers representatives (Italy, Tunisia, Malta), researchers, FAO, MEDAC, NGO (Oceana), Italian DG Pesca officers	To present the MareFrame SoS case study.	MareFrame recognized as relevant to support the management implementation process, enforcing the participation of stakeholders through the assessment work
WP5-WP6 meeting	23-25 June 2015	Oristano ,	MareFrame core team: M. Rahikainen, F. Colloca, M. Sinerchia	Training on structuring a decision analysis for demersal fisheries in the SOS.	A road toward a decision support tool for the SoS CS defined. A first version of the decision tree built.
CS meeting	22-23/06/2016	Palermo	n. 19 participants. Fishers representatives, Vessel owners, NGO (Oceana), FAO MedSudMed officer, MEDAC, Sicily Region officers, CNR staff, MareFrame researchers	Multi criteria analysis (MCA) to support DST	Weight and preferences of MCA criteria and options defined by participants.
FAO MedSud Med project meeting	3-7/10/2016	Rome	FAO MedSudMed officers, researchers for Italy, Tunisia, Malta	Presentatio n of GADGET results	GADGET recognized as an important alternative method to XSA/VPA for the assessment of DPS and HKE. Collaboration with Tunisian researchers for the development of the multispecies GADGET (new stomach contents datasets).
GFCM WGSAD	7-12/11/2016	Rome	FAO and GFCM officers, researchers from Mediterranean countries	Presentatio n of GADGET results	GADGET results and on the stock status included in the stock assessment forms to be submitted to the GFCM-SAC.

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ATLANTIS

The development of the End-to-End ecosystem model ATLANTIS for the Strait of Sicily (SoS) was an interdisciplinary effort and represents the first attempt at systematically gathering data for the Strait of Sicily ecosystem (from bacteria to top predators and fishing fleets). It represents a first step towards the development of decision support tools for EAFM to explore the combined effect of multiple stressors, natural or anthropogenic, on the strait of Sicily ecosystem and societal welfare.

ATLANTIS was built in order to evaluate potential methods and tools (such as ecological indicators) for use in ecosystem-based fisheries management using a Management Strategy Evaluation (MSE) approach, that involves assessing the consequences of a range of management strategies or options and presenting the trade-offs in performance across a range of management objectives.

It is composed by a set of submodels (Fig. 6.2.2). It features a deterministic biophysical submodel, which is spatially resolved in three dimensions using a map made up of polygons and vertical layers. It follows tracks the nutrient flow through the main biological groups found in the marine ecosystem of interest. The primary ecological processes considered in the model are consumption, production, waste production and cycling, migration, predation, recruitment, habitat dependency, and mortality. Lower trophic levels (invertebrates) are modelled as biomass pools (although cephalopods and shrimps may have some age structure), while the vertebrates are represented using an age- and stock-structured formulation (which tracks the condition of average individuals). The physical environment is also represented explicitly - via a set of polygons matched to the major geographical and bioregional features of the simulated marine system. Physical forcing fields (currents, temperature and salinity) are included using results of an external hydrodynamic model. The exploitation submodel allows for multiple fleets, each with its own characteristics (regarding gear selectivity, habitat association, targeting, effort allocation and management structures).

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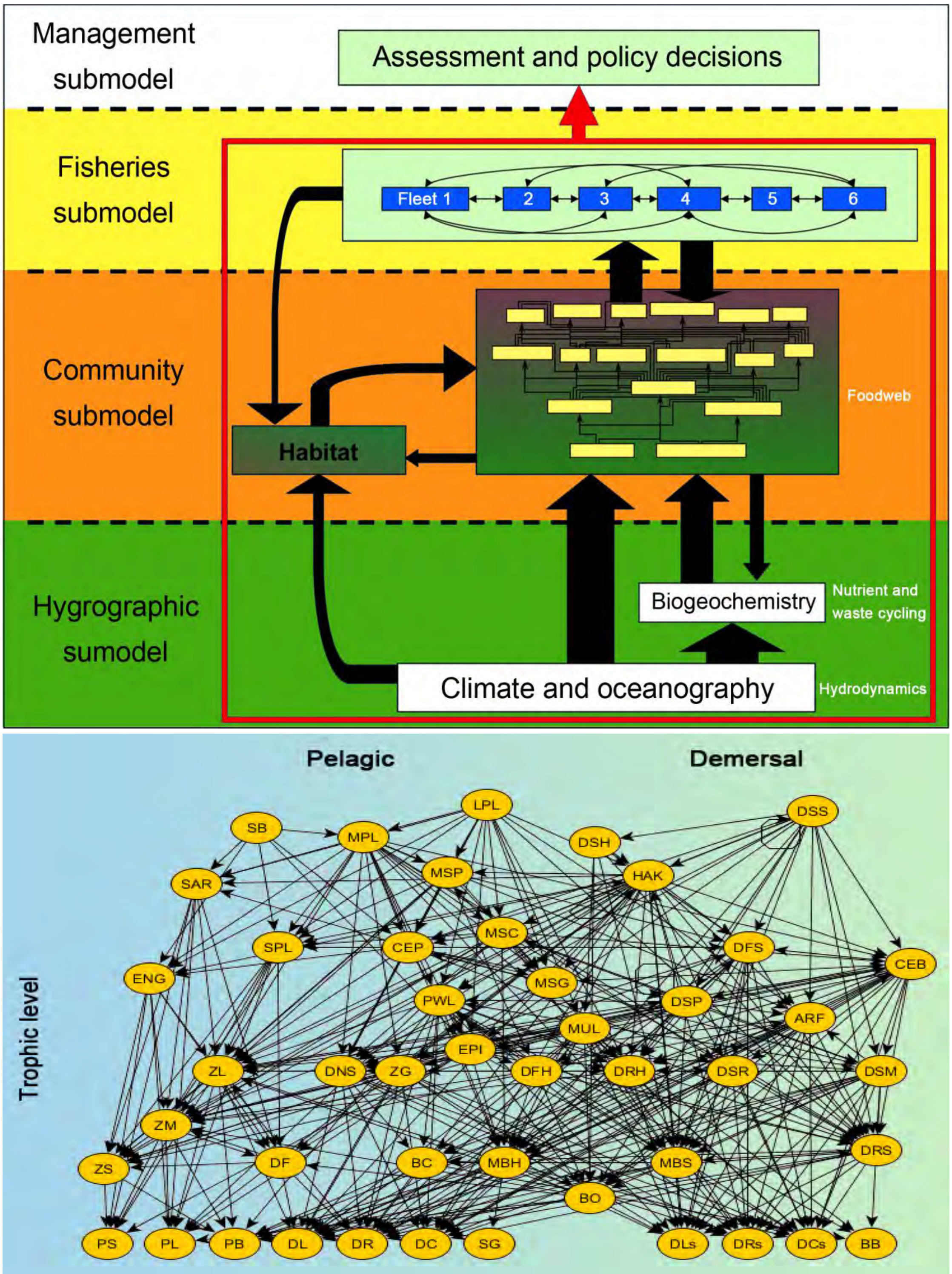

Fig. 6.2.2. Structure of ATLANTIS for the SoS ecosystem

A review by FAO (Plaganyi, 2007) on models for an ecosystem approach to fisheries management ranked ATLANTIS as currently being the most suitable model for allowing EAFM as:

1. It includes the full trophic spectrum;
2. Vertebrates such as fish are modelled using age-structured formulations;

3. Lower trophic level groups are better represented than in most whole ecosystem models, whereas the upper trophic level groups are better represented than in other biogeochemical models;
4. The model is spatially resolved and allows for spatial management scenarios (e.g. MPAs of fishing restricted areas);
5. There is detailed coupling between physical and biological processes;
6. It considers the socio-economic consequences of management scenarios.

The SoS ATLANTIS ecosystem is composed by 58 functional groups, 26 of which are vertebrates. The most important commercial species (anchovies, sardine, red-mullet, hake, giant red shrimp and deep-water rose shrimp) are represented at species level. The primary ecological processes considered are consumption, production, waste production and cycling, migration, predation, recruitment, and mortality. The Sicilian fisheries fleet segments are represented by sub-fleets: two inshore and off-shore bottom-trawlers, pelagic trawlers, large long-liners, small and large purse-seiners, small scale fishery and mixed gears fishery.

The model was developed in order to reproduce the structure and functioning of the ecosystem, with particular focus on commercially important species, to model the effect of climatic or man-induced changes on marine habitats and ecosystem, and use it for scenario testing and trade-offs evaluation related to the application of different fisheries management scenarios on the marine ecosystem and on the socio-economic compartments of the Strait of Sicily.

GADGET

GADGET is an acronym for the “Globally applicable Area Disaggregated General Ecosystem Toolbox” which is a statistical model of marine ecosystems designed to be multi-fleet and capable of including predators and mixed fisheries issues (Beagley and Howell, 2004).

GADGET can be used also for single species assessment and in European waters is currently used to assess stocks in the ICES area (e.g. southern stock of hake in divisions 8.c and 9.a, tusk and golden redfish in Iceland waters). In the Mediterranean was applied for the assessment of hake in GSA 9 (Bartolino et al., 2011).

It is an age–length structured parametric forward simulation model, coupled with an extensive set of data comparison and optimization routines. Processes are generally modelled as dependent on length, but age is tracked in the models, and data can be compared on either a length and/or age scale. GADGET works by running an internal model based on many parameters, and then comparing the data from the output of this model to observed data to get a goodness-of-fit likelihood score. The parameters can then be adjusted, and the model re-run, until an optimum is found, which corresponds to the model with the lowest likelihood score. The GADGET framework consists of three parts: 1) a parametric model to simulate the ecosystem, 2) statistical functions to compare the model output to data, 3) search algorithms to optimise the model parameters.

In SoS area, GADGET is aimed at assessing the effects of prey-predator interactions between hake (HKE), deep-water rose shrimp (DPS) and horse mackerel (HOM), including cannibalism, thus providing a simplified multispecies assessment and management tool for off-shore trawl fisheries. The first two stocks are included in the GFCM management for trawl fisheries in the region plan because they represent two of the most important stocks whereas horse mackerel is the key foraging pelagic species in the region. Recent data on hake stomach contents in the area indicated that HKE largely prey on HOM (up to 60% of the diet in weight depending on hake size class), and significantly on DPS (up to 5% of the diet) and HKE itself (up to 4%). In this regard GADGET will be a key step toward the inclusion of prey-predator interactions in stock assessment. For example, a rebuilding of the hake stock can have a significant impact on the standing stocks of HOM and DPS, through an increasing of natural mortality due to predation, and in turn on fisheries catches and revenues. At the same time multispecies GADGET model will allow to progressively include important ecosystem and population processes as soon as new knowledge will be available, for instance on the effects of climatic change) on growth/recruitment/mortality (i.e relationships between temperature and recruitment. Finally GADGET, has a the potential to develop single species/multi species forecasting of management scenarios to simulate the effects on catches/landings, fishing-mortality at age etc.

Currently the assessments of DPS and HKE stocks in SoS area (GSAs 12-16) are based on the Extended Survivors Analysis (XSA) combining the catch data of the main fleets exploiting the stocks in the area (Italian, Tunisian and Maltese trawlers, small scale vessels) and using the MEDITS bottom trawl survey data as tuning index. Single species GADGET models provide more reliable assessments of fishing mortality and stock biomass of the three stocks than standard XSA assessments. In particular, GADGET allows to model each fleet separately using ad-hoc fleet based selectivity models instead of mixing fleets catch in a single catch at age matrix, as it is currently done for stock assessment in the region. For hake in particular trawl selectivity was modelled in GADGET as a double logistic curve with a right tail, thus acconting for the reduced catchability toward big specimens of Mediterranean trawlers (Bartolino et al., 2011). In addition, GADGET is flexible in parameters estimations for those stock parameters that are more uncertain (i.e growth parameters).

Results of GADGET assessments for DPS and HKE have been discussed during the FAO project MedSudMed and presented during the GFCM working group on stock assessment of demersal species (WGSAD) held in Rome on 7-12 November 2016. The WGSAD recognized the new GADGET assessments as an important step toward a more sound scientific advice on the status of the two stocks and the results will be presented at the subcommitee for the central Mediterranean and the Scientific Advice Committee of GFCM to be considered for the provision of advice.

Detailed presentation of the best models selected

ATLANTIS

Settings and assumptions

The model domain corresponds to the North sector of the Strait of Sicily and includes the FAO-GFCM geographical sub-areas (GSAs) 15 (Malta Island) and 16 (South of Sicily), and it was divided into 37 polygons, 5 of which are islands, matched to the major geographical and bioregional features of the simulated marine system. All input and output data are organised according to the model vertical and horizontal displacement of the polygons in a 37x6 grid format (Fig. 6.3.1).

Each polygon is vertically divided in up to five water layers, depending on the average box depth, and one sediment layer (Fig. 6.3.2).

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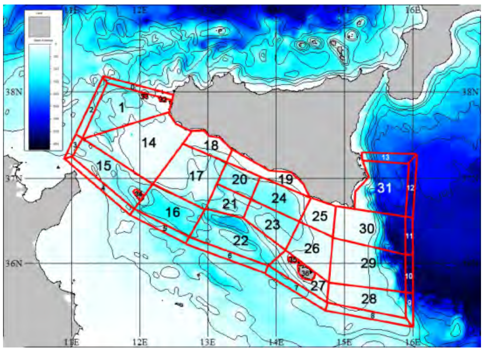

Fig. 6.3.1. The Strait of Sicily bathymetry and model domain containing 37 polygons.

Fig. 6.3.2. Depth layers per box (BB indicates Boundary Boxes which are not explicitly modelled), box 32-36 are islands.

The biological groups included in ATLANTIS were made up of functional groups (aggregate groups of species with similar size, diet, predators, habitat preferences, migratory patterns and life history strategy) and dominant target species in the Strait of Sicily fisheries. The biological community was constructed using mainly timeseries (1995-2012) of density and biomass data observed during the MEDITS survey programme (International bottom trawl survey in the Mediterranean, MEDITS). A total of 1647 species sampled were aggregated into 58 functional groups, 26 of which vertebrates and 32 between plants, invertebrates and detritus groups (Table 6.3.1.). The most commercially important species were represented at species level, including anchovy (*Engraulis encrasicolus*), sardine (*Sardina pilchardus*), hake (*Merluccius merluccius*), red mullet (*Mullus barbatus*), giant red shrimp (*Aristaeomorpha foliacea*), and deep-water rose shrimp (*Parapaeneus longirostris*).

Table 6.3.1. Functional groups. Target species (orange), primary producers (green), detritus groups (gray).

VERTEBRATE GROUPS		INVERTEBRATE GROUPS	
Code	Long Name	Code	Long Name
ENG	*Engraulis encrasicolus*	ARF	*Aristaeomorpha foliacea*
HKE	*Merluccius merluccius*	DPS	*Parapaeneus longirostris*
MUL	*Mullus barbatus*	CBH	Benthic cephalopod shelf
PAG	*Pagellus erythrinus*	CBS	Benthic cephalopod slope
SAR	*Sardina pilchardus*	CPH	Pelagic cephalopod shelf
DFH	Demersal fish shelf crustacean feeders	CPS	Pelagic cephalopod slope

DSM	Demersal fish shelf mixed food	BC	Benthic Carnivore
DSP	Demersal fish shelf piscivorous	PL	Diatom
DSR	Demersal fish shelf rocky	DF	Dinoflagellates
DFS	Demersal fish slope	EUP	Euphausiids
RSH	Demersal rays shelf	ZG	Gelatinous zooplankton
RSS	Demersal rays slope	ZL	Large zooplankton
SSH	Demersal sharks shelf	MA	Macroalgae
SSS	Demersal sharks slope	MBH	Macrobenthos shelf
EPI	Epipelagic fish	MBS	Macrobenthos slope
LPL	Large pelagics	BO	Meiobenthos
MM	Marine mammals	ZM	Mesozooplankton
MPL	Medium pelagics	MB	Microphtybenthos
MSC	Mesopelagic fish slope crustacean feeders	ZS	Microzooplankton
MSG	Mesopelagic fish slope jelly feeders	DNH	Natant decapods shelf
MSP	Mesopelagic fish slope piscivorous	DNS	Natant decapods slope
SPL	Other Small pelagics	PB	Pelagic Bacteria
PSH	Pelagic sharks	PS	Picophytoplankton
TUR	Sea turtles	DRH	Reptant decapods shelf
SB	Seabirds	DPS	Reptant decapods slope
TRA	Trachurus spp	SG	Seagrass
		BB	Sediment Bacteria
		SUH	Suprabenthos shelf
		SUS	Suprabenthos slope
		DL	Labile detritus
		DR	Refractory detritus
		DC	Carrion

The species composition and biomass contribution to each functional group is shown in ANNEX 6.1. Invertebrate groups are represented as biomass pools. Cephalopods and shrimps are divided into juveniles and adults stocks.

Vertebrate groups are represented as age-classes, typically 10. The biomass of functional group is calculated as the sum of the biomass of all age-classes. In addition to these living biological groups, pools of ammonia, nitrate, phosphorus, silica, carrion, labile and refractory detritus are also represented dynamically.

Physical forcing

The outputs from the OPATM-BFM model are used to provide advection, diffusion, temperature and salinity data for Atlantis. Atlantis expects the currents to be in the form of bulk exchanges between boxes/layers. These bulk exchanges were calculated by imposing Atlantis model geometry on the hydrodynamic model and then integrating the currents passing through the faces (horizontal) and layers (vertical) of the model geometry. River discharge rate and nutrient concentrations were taken from the Global NEWS dataset (Fig.6.3.3)
(http://www.marine.rutgers.edu/globalnews/index.htm).
A point-source file was associated to each box containing rivers, giving for each nutrient the inflow from river runoff and from the atmosphere. Temporal variation is not accounted for so that river flow and nutrient levels are assumed to be constant in time.

Fig. 6.3.3. Global NEWS river runoff dataset

Based on these data sets the nutrient input into the different ATLANTIS polygons were extracted on a daily basis and added to the polygons as a so called ‘point source’. Point sources in ATLANTIS contain all externally added substances that are needed to describe and force the ecosystem or to analyse processes.

Besides the river point sources, nutrients (and oxygen or CO_2) are introduced into the model via the atmosphere. A sink of all nutrients is, beside the biological processes in the water column and the upper sediment, the decay and burying of nutrients in the water column or the transport out of the model domain.

Parametrization

Biological parameters

The model includes a large number of parameters that require values including growth rates derived from von Bertalanffy growth parameters, clearance rates, mouth gapes, vertical distribution and migration, stock recruitment relationships, linear and quadratic mortality rates representing habitat limiting factors, viruses and diseases and habitat types and preferences. Data on the maximum biomass observed for any given polygon were extracted from the bottom trawl surveys GRUND (1994-2008) and MEDITS (1994-2013) datasets and adjusted for catchability calculated as the ratio of acoustics derived biomass and biomass measured by catch. Other information on functional groups movements including average swimming speed, minimum and maximum occurrence depth, temperature and salinity range, number, weight and period of migration in and out of the model domain (e.g. tuna) were taken from literature, reports and FishBase (Froese and Pauly, 2000).

Information on length-weight relationships, growth rates for vertebrate and invertebrates, energy allocation to structural vs. reserve weight, respiration rate, vertebrate size at age and length-weight relationship parameters taken from local studies, literature, reports and FishBase.

Estimates of total mortality (M) and fishing mortality (F) are derived from analytical assessments (GFCM and STECF stock assessment working groups). For the stocks where standard assessments are lacking, we derived mortality estimates (Z) from MEDITS data using a catch curve approach. In other cases from FishBase. For cephalopods and non-commercial crustaceans a possible approach to derive Z (P/B) estimates is to use the empirical equation of Bray (1990, 1999a, 1999b, 2001, 2012).

Trophic interactions

The model trophic structure (Fig. 6.3.4) was constructed by combining stomach contents data available for the Strait of Sicily (e.g. red shrimps, hake, anchovy, sardine, Triglidae, etc.) with data from the Gulf of Castellammare (about 30 species) and other Tyrrhenian areas, and with measurements of δ13C and δ13N for 42 species from three different Sicilian coastal areas (Fanelli et al., 2009, 2010, 2011; Sinopoli et al., 2012) in order to define functional groups and assign species accordingly based on their trophic role. For the other functional groups diet data have been gathered from published studies. Field data on consumption rate of Mediterranean fish are available for a set of species (Carpentieri et al., 2006, 2007, 2008). For other species the Q/B has been calculated using empirical equations (Pauly et al., 1990; Palomares and Pauly 1989; 1998). In ATLANTIS feeding interactions are not defined simply based upon the stomach content data but they are parameterized as 'availability' of one prey species to a certain predator. Based on this availability, the biomass of the predator, the biomass of the prey, the spatial overlap (e.g. a pelagic fish that is mainly in the upper water layer will not at all overlap with a benthic species) the efficiency and the clearance rate the gut content can be calculated in ATLANTIS as an output. Doing this calculation backwards one can estimate a basic version of the availability matrix when the remaining parameters are known using the CSIRO "Availability Calculator".

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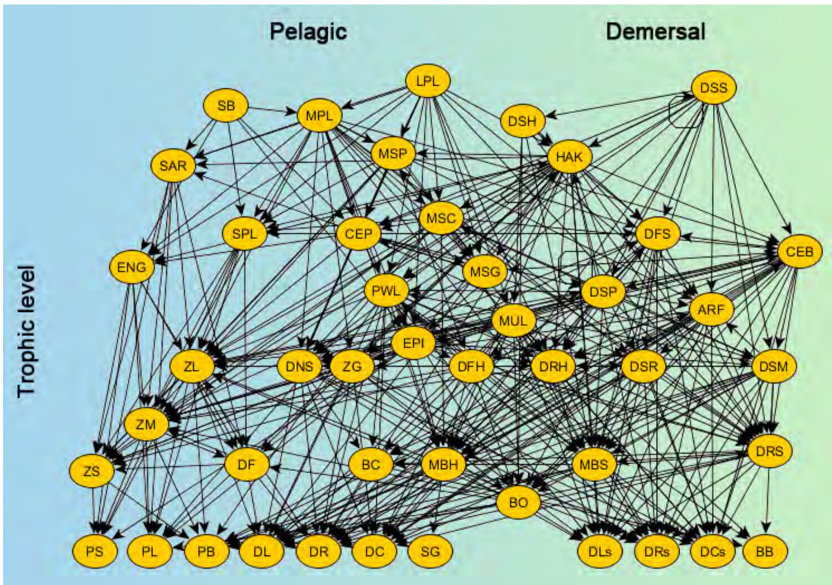


Fig. 6.3.4. Atlantis trophic interactions matrix

All vertebrate groups are split into adults and juveniles (both predator and prey) in the diet matrix, assuming that juveniles are individuals smaller than or equal to the length at 50% maturity (L_m L_m).

Fishing fleets

The top-down control of the ecosystem in the form of fishing pressure was included as multipliers of fishing mortality F by species and fleet using disaggregated landings and discards data from the IREPA database for the years 2004 to 2013. Eight fishing fleets based on gear, mesh size and target assemblage were defined are explicitly represented in the model: bottom trawlers, pelagic trawlers, pelagic artisanal vessels, demersal artisanal vessels, purse seine, demersal longline, pelagic longline, gillnets and trammel nets.

Model fitting

The calibration of the model was carried out in 3 phases:

- full community ecosystem without explicit fishing fleets;
- full community ecosystem with explicit fishing fleets imposing a constant fishing mortality, the average value over the last 5 years,
- full community ecosystem with explicit fishing fleets and time varying fishing mortality

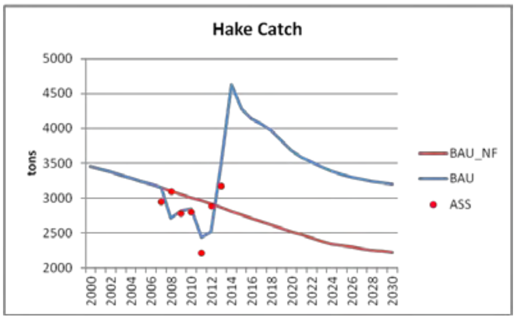
During the first phase, the simulations were carried out considering all biological communities, but without the fishing fleets as forcing. This was compensated by an increase in natural mortality for the various groups. During this phase, the calibration of the model was focused on the coexistence of the various groups and the reproduction of the main biological characteristics simulated: size, growth rates, mortality, diet and predation.

Once the ecosystem was balanced in terms of functional groups coexistence, biomass within range of observations, sizes and growth rates and stable dynamics of the groups, the second stage of calibration involved the explicit development of the fleets of fishing vessels:

1.	Trawling coastal coastal bottom trawl	cstOTB OTB-24
2.	Trawling offshore deep-sea bottom trawl	offOTB OTB-40
3.	Fishing small scale fishery (fixed gears)	smaSF
4.	Longlines large pelagics	lonLP
5.	Pelagic trawl pelagic trawl	pelOTB
6.	Purse seine large pelagics (tuna)	purSL
7.	Purse seine small pelagics (sardine + anchovy)	purSS
8.	Mixed gears	mixG

Description of the characteristics of the fishing fleets is provided in D.5.1 and D 5.2.

The third phase of the calibration included was dedicated to the fine tuning of the fisheries module. In particular, during this phase of model calibration, for hake annual estimates of fishing mortality have been incorporated in the model as forcing to the target species (Figs 6.3.5 and 6.3.6).



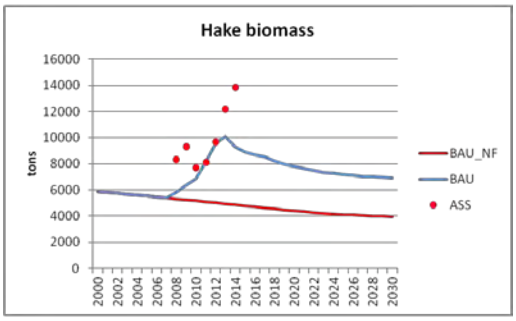


Fig. 6.3.5. The effect of introducing direct F timeseries for hake biomass and landings. BAU: Business As Usual with F timeseries. BAU_NF: Business As Usual with constant F

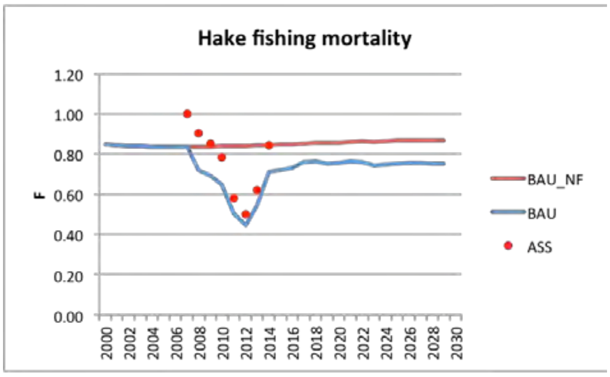


Fig. 6.3.6. Output of F mortality. BAU: Business As Usual with F timeseries. BAU_NF: Business As Usual with constant F.

GADGET

Conceptual model

GADGET in SoS is designed to model interactions between 2 fish populations (hake HKE and horse mackerel: HOM), 1 shrimp population (deep water rose shrimp 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. 6.3.7). 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.

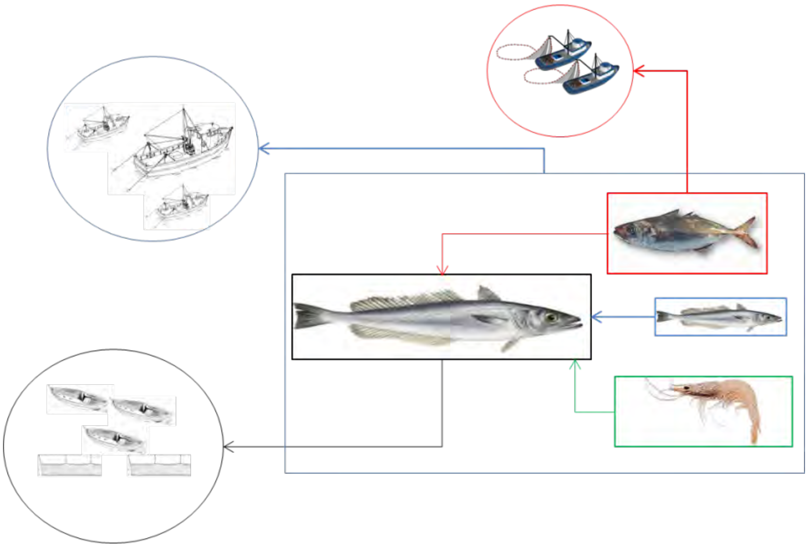


Fig. 6.3.7. Conceptual structure of the multispecies GADGET model for hake (HKE), horse mackerel (HOM) and deep water rose shrimp (DPS), with HKE predator of HOM, DPS and HKE (cannibalism). Fleets exploiting the three stocks: a) bottom trawlers; b) purse seiners and mid-water trawlers, c) artisanal vessels.

Settings and assumptions

HKE and HOM populations are defined by 2 cm length groups, whilst DPS by 2 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 and third 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 range in between 0 and 4, this latter used as plus group. Recruitment take place in the second and third quarter. 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) ad hoc implemented in R (see ANNEX 6.3).

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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 have a reduced trawl catchability (see Bartolino et al., 2011) a new double logistic function was developed. It assumes a dome shape, but with a constant (at some level) right tail, to reproduce that over a certain size the catchability decreases with fish size up a constant level.

The new function is the following:

$a_l,\ a_r,\ l_{50},\ r_{50} > 0,\quad l_{50} < r_{50},\quad 0 \le p \le 1,\quad L > 0$ 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. 3.8)

In the above formulation, parameters and play the same role in the right tail as the corresponding parameters andfor the left side, while *p* indicates the proportion of fish captured after length (Fig. 6.3.8)

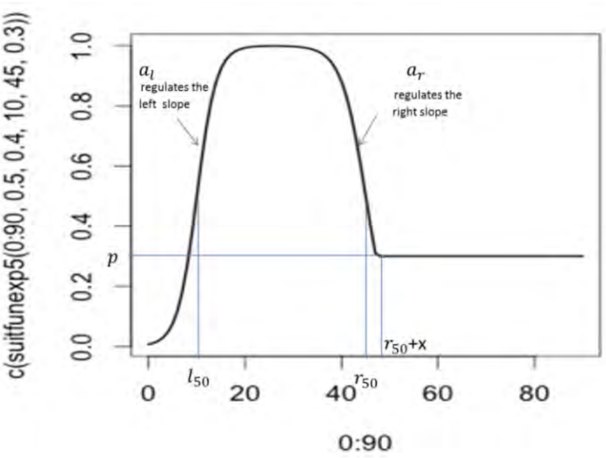

Fig. 6.3.8. The dome-shape and constant right tail selectivity function assumed for modelling the hake capture by bottom trawl fleets.

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

Prey consumption rate

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):

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

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":

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

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 calcutaled 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:

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

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∞ 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 (Temmings and Herrmann, 2009). For hake in SoS we fixed a/b =0.8.

The relationship in Fig. 6.3.9 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:

$ML=m_0L^{m3}$ (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).

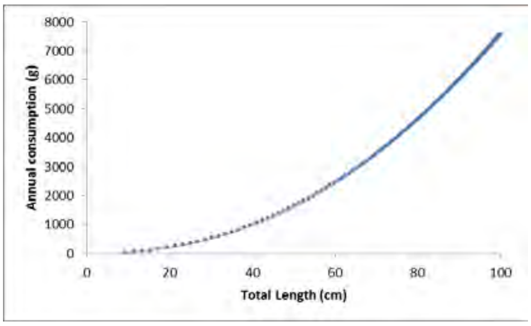

Fig. 6.3.9. Annual consumption curve for hake in the length range between 4 and 100 cm TL

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_2. To account for a differtial 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:

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

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* (Fig. 6.3.10).

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

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.

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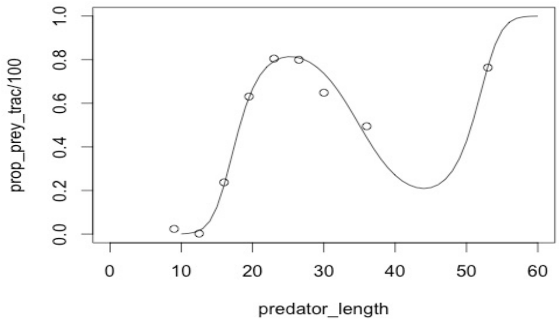

Fig. 6.3.10. Observed consumption for hake vs the predator length, with overimposed third degree polynomial curve from a beta regression model. The consumption is measured by the prey/predator length ratio, on the ground of the horse mackerel, the most consumed prey.

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. 6.3.11 a) to shift the suitability function (Fig. 6.3.11 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. 6.3.11 a).

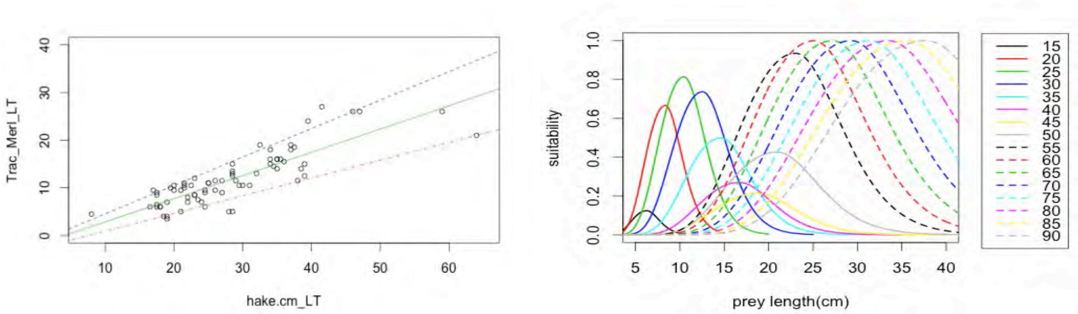

Fig. 6.3.11 a) Prey length – predator length relationship for hake (HKE) and horse mackerel (HOM) in the SoS area. b) Modified Andersen & Ursin (1977) suitability function, expressing the suitability of HKE and HOM in terms of length, for different HKE lengths.

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. 6.3.12). Function is now created ad hoc to resemble the data variation:

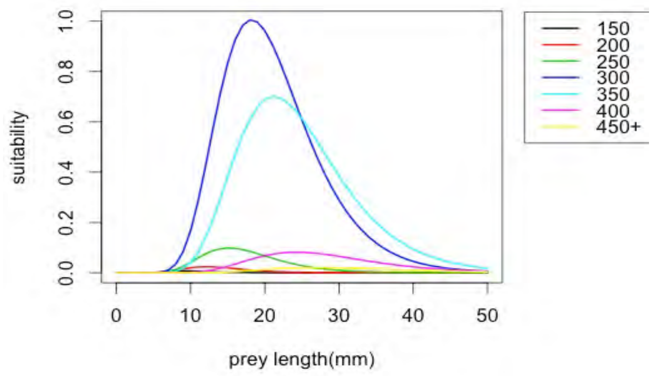

Fig. 6.3.12. Modified Andersen & Ursin (1977) suitability function, expressing the suitability of DPS in terms of length, for different HKE lengths.

The datasets included in the GADGET SoS model and the relative contribution to the final total likelihood are showed in Table 6.3.2.

Table 6.3.2. 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

Parametrization

ANNEX 6.3 lists fixed and estimated parameter values from both the single- and multi-species GADGET for all the species analysed.

Model fitting (best model)

The implementation of the Gadget multi-species model has implied the parametrization of single species models which in a second modelling step were linked by trophic interactions as well as interactions with fleets. For each input data set a specific likelihood function was used to compare the model output to the data during the estimation. Each likelihood component calculated a likelihood score for that individual component. 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. We also made use of visual inspection criteria, in order to exclude implausible results from models supposedly estimated under strong parameter correlation, which can conduct to multiple optimal solutions. Annex 6.4 reports the following results of model fitting for single/multispecies models:

- 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 for both single species models;
- iv) residuals.

Single species model for hake (HKE)

Hake stock biomass, catch, fishing mortality (F_{bar} at age 1-6) and recruitment are showed in Fig. 6.3.13 – 6.3.14. The annual catch has decreased from 2002 to 2011 from 4000 to 2300 t. Biomass is increasing since 2007 and annual recruitment show fluctuactions between 30 and 80 million. Fishing mortality was stable between 0.4 and 0.5 to increase in 2012-2014 ut to 0.7. The assumed F_{MSY} (F_{01} as proxy of F_{MSY}) is 0.18, thus the stock has been in overexploitation in the last years.

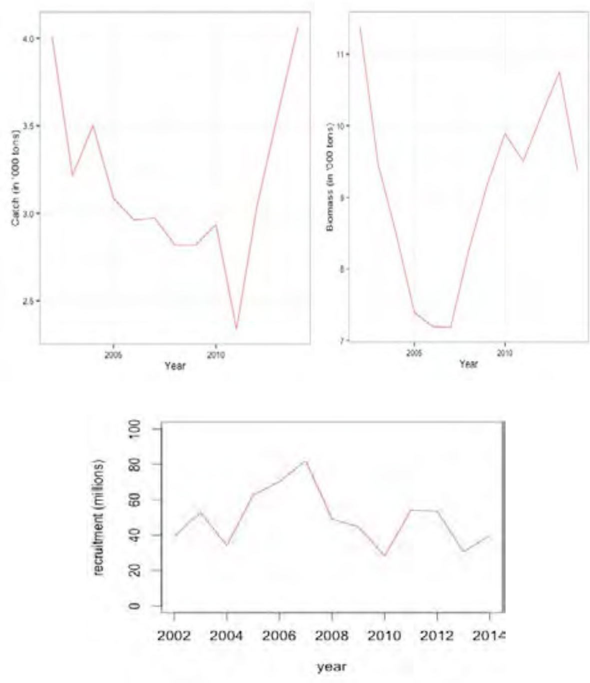

Fig. 6.3.13. Single species GADGET model for HKE: catch, estimated biomass and recruitment

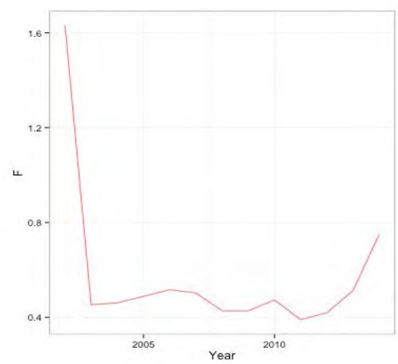

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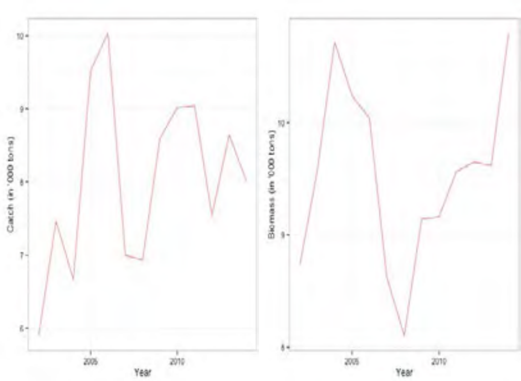
Fig. 6.3.14. Single species GADGET model for HKE: Fbar(1-6).

Single species model for deep-water rose shrimp (DPS)

DPS stock biomass, catch, fishing mortality (Fbar at age 1-3) and recruitment are showed in Fig. 6.3.15 – 6.3.16. Annual catch range between 6000 and 10000 t. Stock biomass showed large fluctuactions from 13000 t (2004, 2014) to about 8000 t in 2008. Fishing mortality ranged between 1.2 and 1.7 in the period 2005-2014. The assumed F_{MSY} (F_{01} as proxy of F_{MSY}) is 0.85, thus the stock has been in overexploitation in the last years.



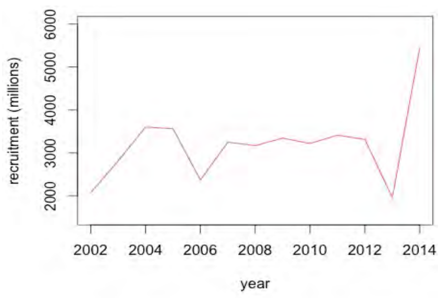


Fig. 6.3.15. Single species GADGET model for DPS: catch, estimated biomass and recruitment

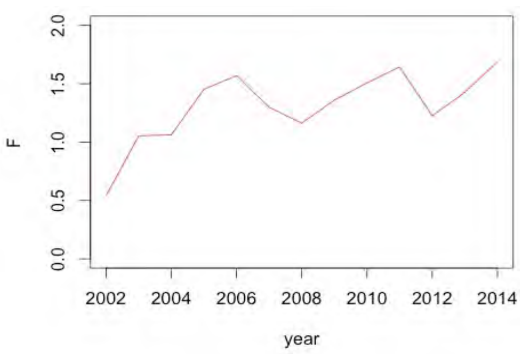


Fig. 6.3.16. Single species GADGET model for DPS: Fbar(1-3) and recruitment.

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Single species model for horse mackerel (HOM)

HOM catch show a reduction since 2002, from 8000 to 2900 t. Stock biomass fluctuate between 9000 and 17000 t (Fig. 6.3.17). Fishing mortality declined from 0.8 to 0.19 in 2014 (Fig. 6.3.18).

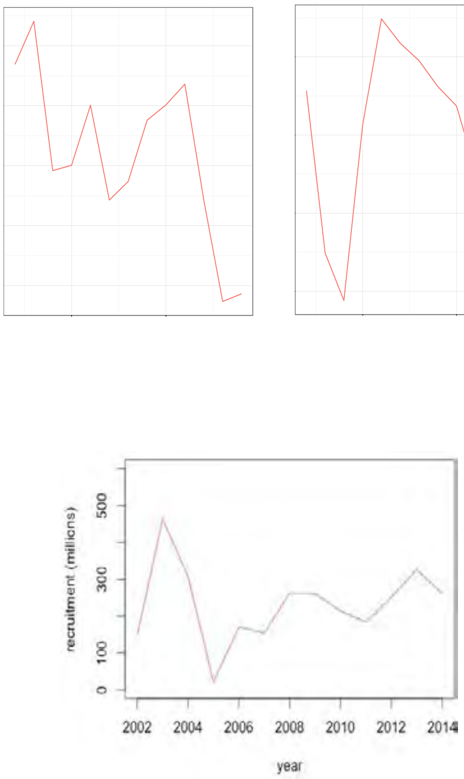

Fig. 6.3.17. Single species GADGET model for HOM: catch, estimated stock biomass and recruitment.

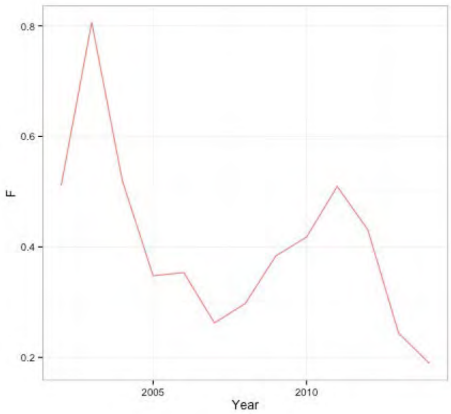

Fig. 6.3.18. Single species GADGET model for HOM: estimated Fmean(1-5)

Multispecies GADGET model

GADGET multispecies model for SoS demersal fisheries exploitind DPS and HKE is based on the following prey-predator relationships: HKE prey upon DPS, HOM and HKE juveniles (cannibalism), and other food. The amount of other HKE food was assumed as constant over time. Growth and survival of HKE is not influenced by its prey consumption, within the present model. Thus, prey consumption has no direct effect on predator dynamics, except that the level of cannibalism by HKE can affect mortality of HKE of the 0-2 age groups.

Results in terms of estimated recruitment (Fig. 6.3.19), total biomass (Fig. 6.3.20), fishing mortality (Fig. 6.3.21-6.3.22) are compared with estimates from single species models in Figs 6.3.23-6.3.26.

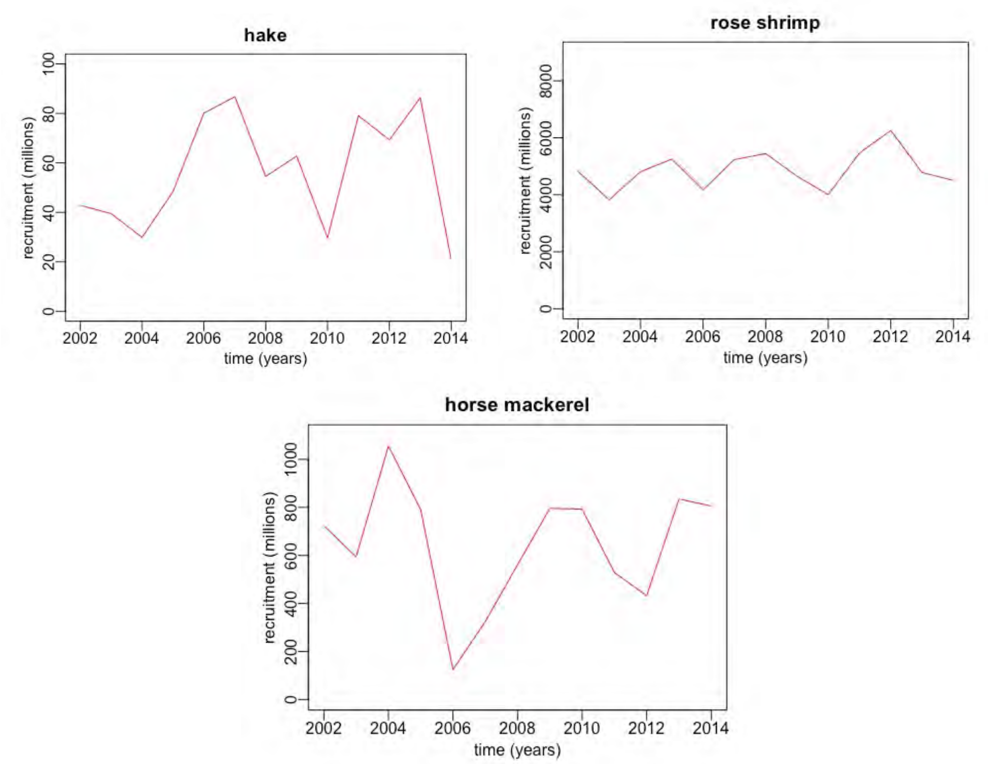

Fig. 6.3.19. Multispecies GADGET Model: estimated recruitment for HKE, DPS and HOM.

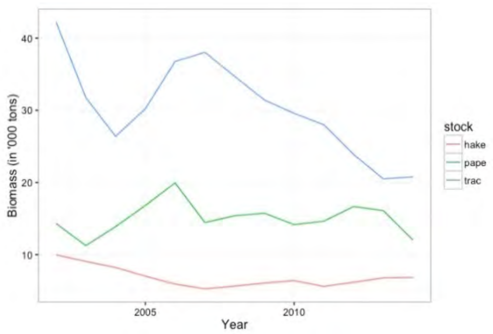

Fig. 6.3.20. Multispecies GADGET Model: estimated biomass for HKE, DPS and HOM.

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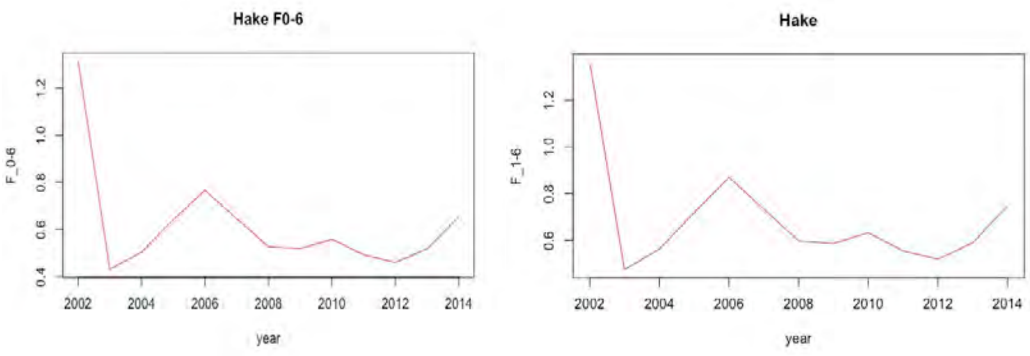

Fig. 6.3.21. Multispecies GADGET Model: estimated mean fishing mortality for hake at ages 0-6 (left graph) and 1-6 (right).

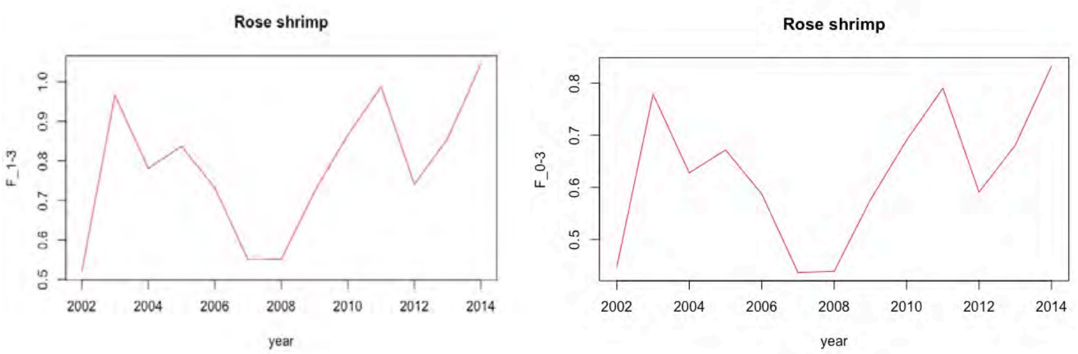

Fig. 6.3.22. Multispecies GADGET Model: estimated mean fishing mortality for rose shrimp at ages 0-3 (left graph) and 1-3 (right).

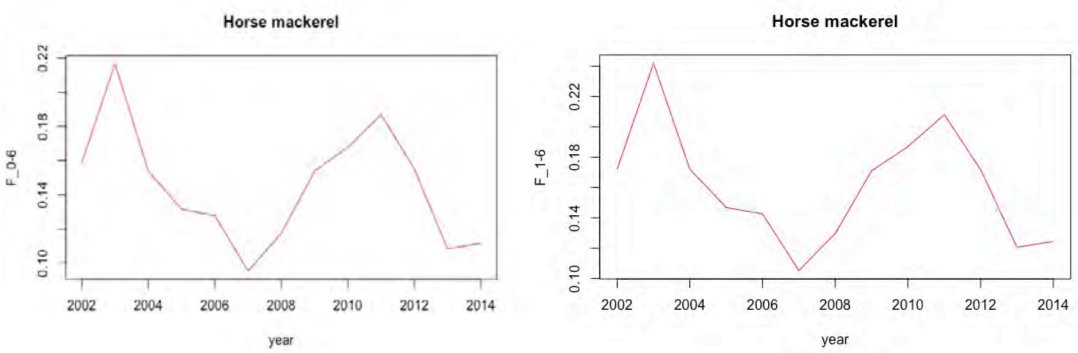

Fig. 6.3.23. Multispecies GADGET Model: estimated mean fishing mortality for horse mackerel at ages 0-6 (left graph) and 1-6 (right).

Comparison single- vs multi-species

In general, the addition or removal of predator-prey interactions resulted in change to population estimates (see Fig. 6.3.24 - 6.3.25). It appears that by combining the three single-species models the optimizer is forced to find a consesus overall-liklihood score, and this can greatly affect the population estimates. This effect was also observed by Trenkel et al., (2004) in their multispecies model for cod and its prey in the Celtic Sea. In general, the temporal trend observed in single species models is kept also in the multispecies model although the predator bomass (HKE) appears lower and the prey biomass (DPS and HOM) higher, probably as effect of the imposed annual consumption. Parameters estimation (e.g. von Bertalanffy k) remained similar in the multispecies model formulation to those in the single species models (see ANNEX 6.4). The impact of HKE consumption on prey population estimates appears particularly relevant for horse mackerel, which is the main HKE prey, with HOM estimated biomass raising of about three times in the multispecies framework. The HOM recruitment trend appears also not in line with that estimated in the single species model.

As a result, the multispecies model returns lower annual fishing mortality of HOM and DPS than those estimated in the single species models and a slightly higher F for HKE (Fig. 6.3.26 – 6.3.27).

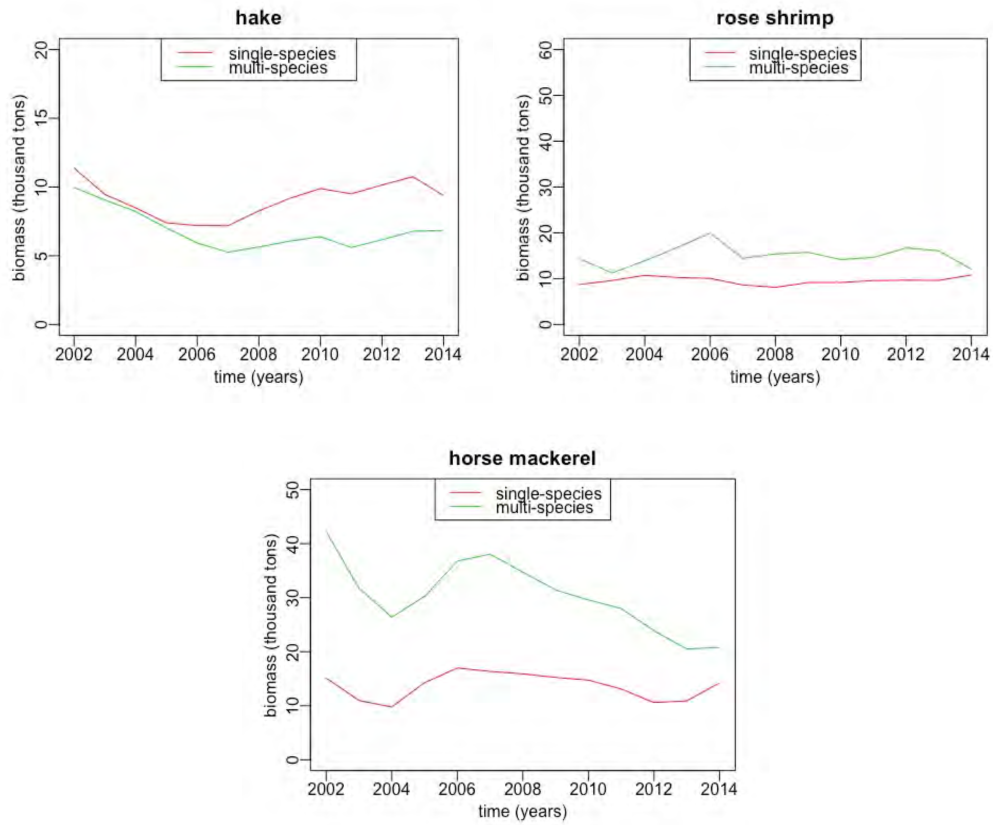

Fig. 6.3.24. Single- vs multi-species GADGET model: biomass comparison.

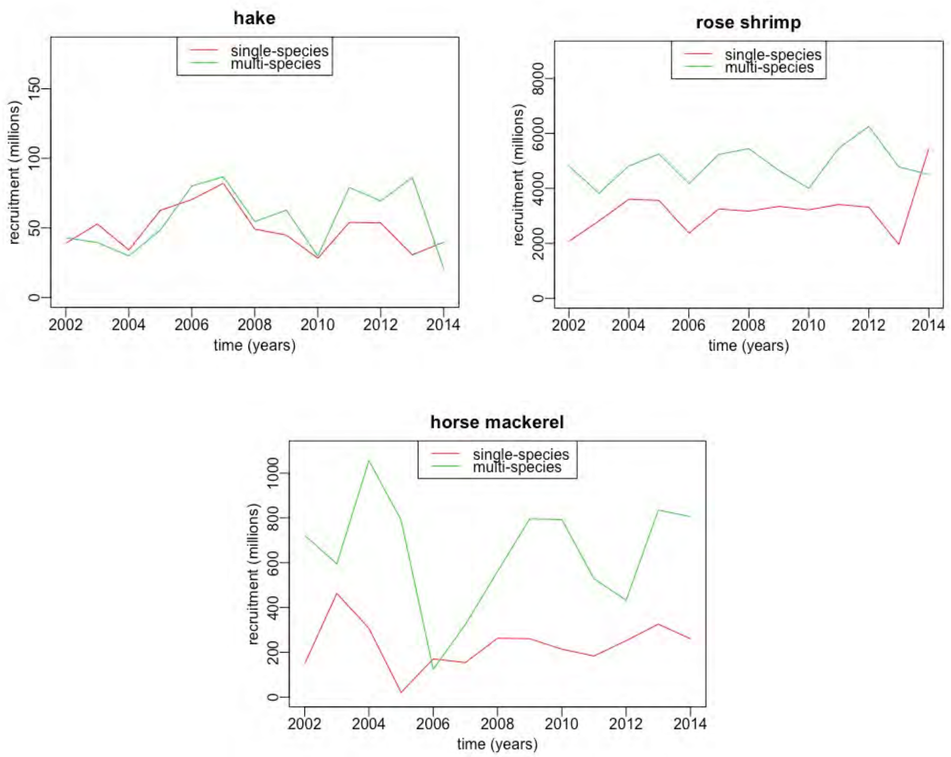

Fig. 6.3.25. Single- vs multi-species GADGET model: recruitment comparison.

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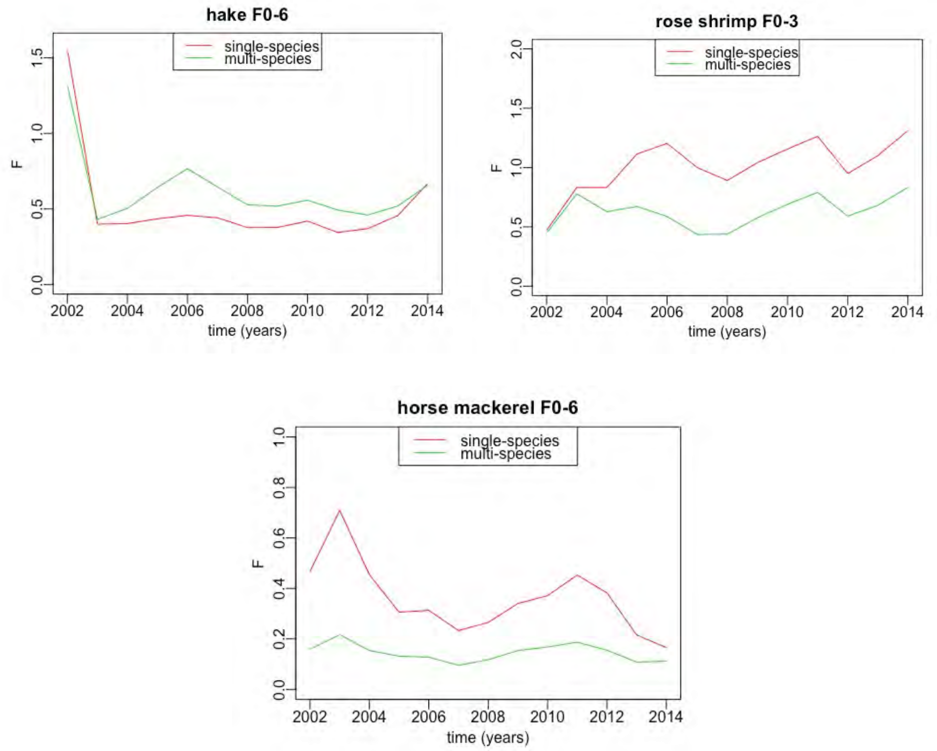

Fig. 6.3.26. Single- vs multi-species GADGET model: fishing mortality.

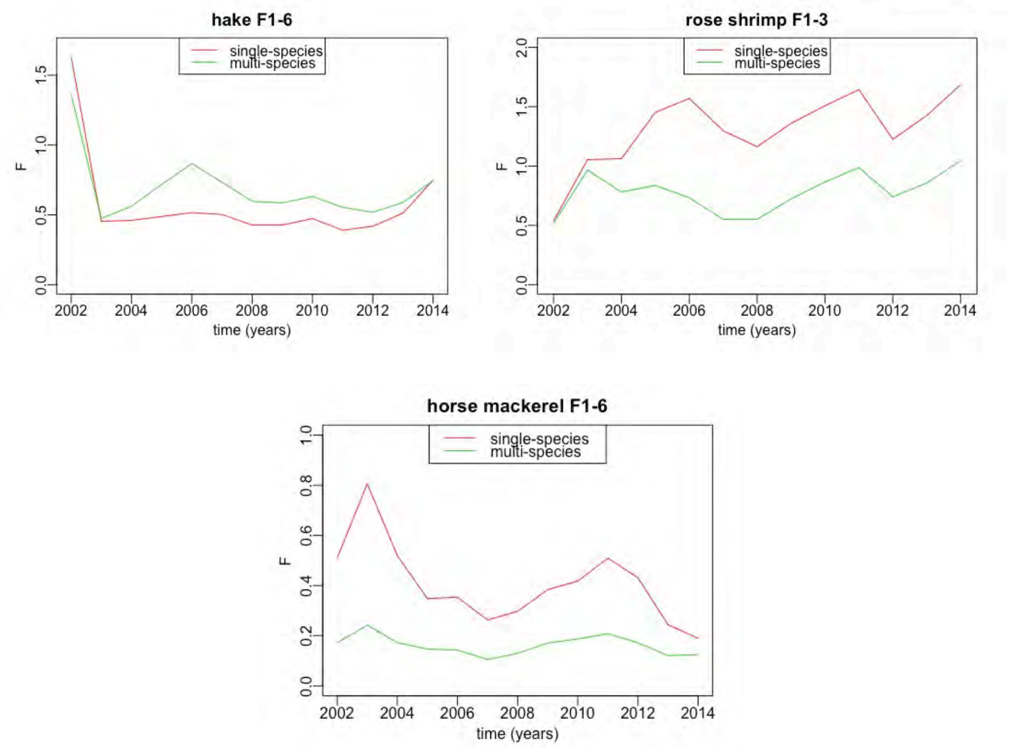

Fig. 6.3.27. Single- vs multi-species GADGET model: fishing mortality

Simulation of management scenarios

Description of scenarios

Management scenarios are based on the application of a set of different technical management measures. ATLANTIS and GADGET are used to simulate the effects of the identified management measures on the achievement of F_{MSY} for DPS and HKE within 2020 as well as on the economic performance of the fleets and the status of relevant ecosystem components.

Table 6.4.1 shows the structure of scenarios tested with ATLANTIS and GADGET. Four main types of management scenarios are developed: i) business as usual; ii) area closures; iii) effort reduction; iv) improved gear selectivity. The management measures assessed include spatial restrictions, fishing days reduction, adoption of more selective trawl nets, development of trade marks/ecolabelling and finally regulated access to fishing grounds (i.e. adoption of n authorized list of vessels). ANNEX 6.2 lists the main data included in the simulations of management scenarios.

In the business as usual (BAU) scenario, both the current exploitation pattern and the status of the ecosystem are kept constant. The other scenarios focus on a main management action (level +++ in Table 6.4.1), with the other measures kept constant (+) or implemented at an intermediate level (++). All scenarios will simulate a fixed number of vessels fishing in the area as effect of the adoption of an authorized list of vessels in the management plan and the improvement of the control and surveillance system (i.e. no IUU fishing taking place).

An initial idea was to add scenarios based on climate forcing as depicted by IPCC scenarios and summarized by the EU project VECTOR using ATLANTIS. However since stakeholders did not perceive climate change as a priority issue for the SoS ecosystem, the modelling work will focus on the other scenarios identified.

Both the ATLANTIS and GADGET models for the SoS ecosytem are used to develop simulations. GADGET is mostly devised to provide more accurate short term simulations on the two main target stocks (HKE and DPS) and ATLANTIS to depict the long-term effects on the ecosystem. The two model differ substantially in their capability to simulate the effect of the identified management measures mostly as effect of their internal structure and simulations functions (e.g. spatial components, functional groups, etc.).

A set of preliminary simulations with ATLANTIS were developed and described in the next sections. In the case of GADGET, scenarios simulations were preliminary produced to simulate a reduction in harvest rate assuming a unique recruitment season during the year. More complex simulation were delayed to the necessity to implement a new simulation function to account for multiple recruitment events during the year observed for HKE and DPS.

Table 6.4.1. Management scenarios simulated in the SoS case study. +: business as usual; ++: intermediate level of the measure implemented; +++: high level of measure implementation.

Measures	**Business as usual**	**Fmsy for deep-water rose shrimp / hake**		
		Area closure based	Effort reduction based	Gear selectivity based
Spatial restrictions	+	++, +++	+, ++	+, ++
Effort reduction (days at sea)	+	+, ++	++, +++	+, ++
Gear selectivity	+	+, ++	+, ++	++, +++
Ecolabelling and certification	+	+, +++	+, +++	+, +++
Regulate access to fishing grounds	+	+++	+++	+++

The simulation work will fuel the development of the DST (decision support tool) in WP6. DST prototype I will be based on the ATLANTIS simulation whereas GADGET outputs will be integrated into the final version of the DST (DST prototype II). In a first stage only simulations based on single measures (e.g. areas closure, change in gear selectivity, F reduction) are developed to be complemented with a more complex simulation approach for the development of DST prototype 2 (Table 6.4.2).

Table 6.4.2. Development of scenarios simulations in the SoS CS and link with the DST development in WP6.

Scenarios simulations	**MODEL**	**DST I**	**DST II**
Single measures	ATLANTIS	X	X
Multiple measures	ATLANTIS, GADGET		X
Climate forcing	ATLANTIS	X	

Indicators

The following sets of indicators were selected to assess the effects of alternative managemen scenarios:

<u>Stocks indicators (ATLANTIS and GADGET)</u>

- Landings: total landings; age composition of the landings, CPUE (ATLANTIS; GADGET);

- Discards: (based on the proportion of the 0 group in the simulated catches; ATLANTIS; GADGET);

- Fishing mortality (F);

- Total biomass;

- Spawning stock biomass;

Socio-economic indicators (ATLANTIS and GADGET)

- Total fisheries revenues: R (ATLANTIS; GADGET);

- Total fisheries costs: C (ATLANTIS; GADGET);

- Total fisheries profits: P, where the profit (P) is: P=R-C (ATLANTIS; GADGET);

- Days-at-sea: assumed as social indicator correlated to the quality of life of fishers (ATLANTIS, GADGET);

- Average salary (based on the impact of the measures on the crew profits wich in turn depends by the work contract ("alla parte") contract (ATLANTIS)

Ecosystem indicators (ATLANTIS)

- Total exploitable biomass (sum of fish, decapod crustaceans and mollusks cephalopods biomass).

- Biomass of forage fish (sum of the biomasses of anchovy, sardine, other small pelagics and horse mackerels)

- Pelagic to demersal fish ratio (the total biomass of pelagic fish groups divided with the total biomass of the demersal fish groups);

- Ratio of of piscivore biomass to benthivor e and planktivore biomass;

- Modified species richness (the fraction of functional groups that have biomass that drops below 50% of the initial total biomass each year, divided by the total number of Atlantis model groups. From Masi et al., 2016)

- Shannon-Wiener diversity index (Calculated as the standardized −(Bi) multiplied by the natural log of B, where Bis the biomass (tons) for each functional group. Index is a unitless measure of system entropy. From Masi et al., 2016).

Business As Usual (BAU)

This scenario assumes that the Strait of Sicily management and environmental forcing will not change in the future years. ATLANTIS simulations are run using the initial conditions and forcing files described in D 4.2. The simulation time starts in 1965, in order to allow enough time for the ecosystem to stabilize to an annual attractor from the initial condition errors. Outputs from the model are analysed from year 2000. Fishing annual mortalities (F) imposed on the stock (Fig. 6.4.1) were taken from assessment estimates for the period available (yellow area). Before this period (blue area) F are assumed as the average of the first 2 years of the assessment timeseries. In the BAU scenario fishing mortality is assumed to be constant at present level (e.g. ~ 0.8 for hake).

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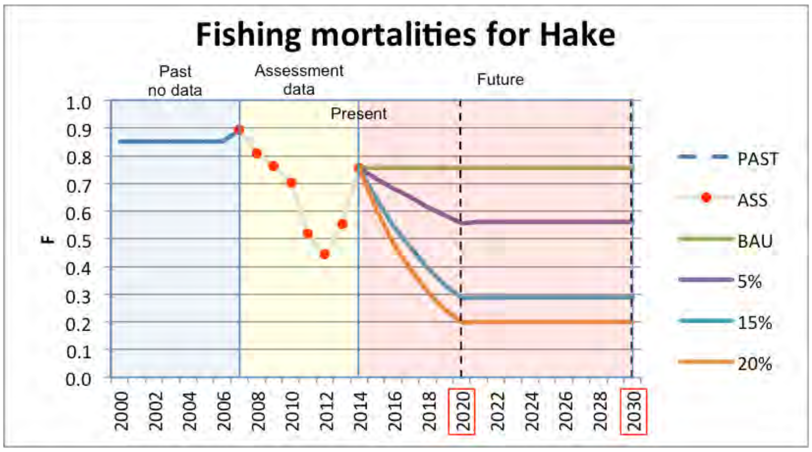

Fig. 6.4.1. Imposed fishing mortality on hake under different management scenarios

GADGET simulations will be based on the last year fishing mortality at age for HKE, DPS and HOM, mean recruitment in last three years, SSB in the last year.

F_{MSY} estimation

The BAU scenario in ATLANTIS was used as baseline over which simulations were re-run by just changing the fishing mortality for hake and deep water rose shrimp to reconstruct the sustainable yield curve and compare it to assessments estimates. The F mortality was progressively increased from 0 to 3.0 for deep water rose shrimp and from 0 to 1.0 for hake to estimate the F_{max}, F_{01} for both stocks. The F mortality was kept constant during the whole period of each simulation.

GADGET inputs and outputs (M-at-age, F-at-age, etc.) are used to estimate F_{01} using the FLR package.

F reduction scenarios

This scenario simulates the effect of the implementation of the Common Fisheries Policy (CFP) for achieving a sustainable exploitation. Member States are now committed to restore and maintain fish stocks above biomass levels capable of producing maximum sustainable yield (B_{MSY}). In order to reach that they must set fishing limits according to the exploitation rate consistent with this aim (F_{MSY}) by 2020 at the latest for all commercial stocks.

The focus of this investigation is to investigate the short-term (2020) and long-term (2030) effect of reduction of fishing from its current level (F_{curr}) to the level necessary to achieve maximum sustainable yield (F_{01} as proxy of F_{MSY}) for hake and deep water rose shrimp, which are target of the same fishing fleets.

In particular, three scenarios of progressive decrease of fishing mortality from 2015 to 2020 have been simulated in ATLANTIS (Table 6.4.3):

1. F reduction from F_{curr} to F_{MSY} for deep water rose shrimp
   5% annual decrease in fishing mortality

2. Intermediate F reduction
   15% annual decrease in fishing mortality

3. F reduction from F_{curr} to F_{msy} for hake
   20% annual decrease in fishing mortality

Table 6.4.3. Fishing mortality reduction to Fmsy levels by 2020 scenarios. Fcurr is shown in yellow and Fmsy in green.

	Hake			Deep water rose shrimp		
	5%	15%	20%	5%	15%	20%
2014	0.76	0.76	0.76	1.1	1.1	1.1
2015	0.72	0.64	0.61	1.05	0.94	0.88
2016	0.68	0.55	0.48	0.99	0.79	0.7
2017	0.65	0.46	0.39	0.94	0.68	0.56
2018	0.62	0.39	0.31	0.9	0.57	0.45
2019	0.59	0.34	0.25	0.85	0.49	0.36
2020	0.56	0.29	0.20	0.81	0.42	0.28

A preliminary set of simulation was also done with GADGET assuming a progressive reduction in the harvest rate of exploitable biomass.

Fisheries Restricted Areas scenario (FRA)

The aim of this scenario is to investigate the effect associated with the closure of three FRAs included in the GFCM management plan and calculate the impact on the two target stocks (HKE and DPS) and fisheries in the medium term (2020) and long term (2030). The institution of the 3 FRAs in the Strait of Sicily (Fig. 6.4.2) was simulated by mean of a reduction of fishing mortality proportional to the FRA percentage coverage of Atlantis polygons (Table 6.4.5). Fishing mortality is redistributed to the surrounding boxes. A more complex simulation is however required to address more consistently the FRAs effects on target stocks and fisheries, as describt in the next sections.

Background

The recent GFCM recommendation 40/2016/4, establishing a multiannual management plan for the fisheries exploiting European hake and deep-water rose shrimp in the Strait of Sicily (GSA 12 to 16), includes the institution of three fishery restricted areas (FRAs) where trawl activities are forbidden (Fig. 6.4.2).

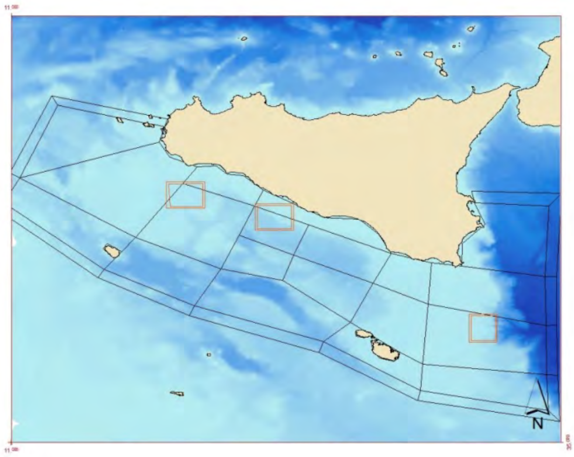

Fig. 6.4.2. Map showing the position of the three Fisheries Restricted Areas (FRA, red boxes) established in the Strait of Sicily by the GFCM recommendation 40/2016/4. The ATLANTIS model spatial grid is also showed.

The main aim of the three FRAs is to protect juveniles of hake (HKE) and deep-water rose shrimp (DPS) when they aggregate in their nursery areas at the end of the planktonic larval life. Nursery grounds of the two species have been proved to be stable over time (Fiorentino et al., 2003; Garofalo et al., 2011; Colloca et al., 2015) thus providing a fundamental prerequisite for their incusion in a conservation network. Issues related to the application of FRAs as management tools to reduce the fishing mortality on juveniles was analysed by Russo et al. (2014) and widely discussed during SOS case study meetings, held in Mazara del Vallo and Palermo in 2015 and 2016. Fishers and fishers representatives generally agreed that spatial closures can ben effective tools to reduce by-catches of juveniles and improve the exploitation pattern of trawl fleets.

Setting the FRAs scenarios

In task 5.3 we defined 2 different area closure scenarios (FRAs scenario) to assess the impact of the implementation of the three FRAs on target stocks and trawl fisheries using ATLANTIS. The first one is a simplified scenario which does not account for stock demography, spillover effects and spatial distribution of fishing effort. The second one is a more complex scenario where the impact of FRAs on the stocks is modelled considering the spatial differences in the catch composition by age group and including both a spillover effect (i.e. juveniles surviving into the FRAs migrate in adjacent areas) and a spatial redistribution of the trawl effort /fishing mortality (Table 6.4.4). In both scenario we assumed a full commitment of local fleets to the FRAs (i.e. no illegal catch). The first scenario will be provided for DST I whereas the second one will be available for DST II.

Table 6.4.4. FRAs scenarios for the SoS case study

Scenario	Closed areas	Data included	Indicators
1	3 FRAs	Catch or fishing mortality in FRAs as proportion of catch in the Atlantis boxes where the FRAs are located. F-at-age in each Atlantis box.	- HKE, DPS stocks: Fbar, SSB - FLEETS: Annual landings, CPUEs, Profits - ECOSYSTEM: Biomass of key functional groups
2	3 FRAs	Catch in FRAs of age 0 HKE and DPS. Catch composition by age group of HKE and DPS by ATLANTIS box. Fishing mortality at age by ATLANTIS box. VMS data on trawl fleet effort in 2015 and fishing effort displacement in the ATLANTIS domain. Estimates of the effort displayed into the FRAs. Data on catch composition and amount by age group of DPS and HKE in each ATLANTIS box. Data on ontogenic migration pattern from nurseries areas (spillover) based on MEDITS survey data.	- HKE, DPS stocks: Fbar, F-at-age, SSB - FLEETS: Annual landings, CPUEs, Profits. - ECOSYSTEM: Biomass of key functional groups.

Modelling data for FRA scenarios implementation

FRAs in the model domain

The proportion covered by the estabished FRAs in the ATLANTIS boxes where they are locates is showed in Table 6.4.5.

Table 6.4.5. Fisheries Restricted Areas area and their percentage coverage of Atlantis polygons

FRA	Box	% FRA	FRA Area (Km^2)
A	14	2.2	145.8
A	17	13.8	629.7
A	18	7.2	95.6
B	19	19.4	326.2
B	20	40.9	633.7
C	29	12.0	646.8
Total			2477.8

Fishing effort

Vessel Monitoring System (VMS) data for 2015, as provided by the Data Collection Framework, were used to calculate a fishing effort index as total number of fishing hours of the trawl fleet in 2015 by each 1 km^2 grid cell of the area (Russo et al., 2015). The proportion of fishing effort displayed in the three FRAs (182.917 hours) was **4.3% of the total effort for the area (**7.830 hours, Fig. 6.4.3a). Effort data are converted into the total amount of fishing pressure observed in 2015 in each ATLANTIS box using GIS (Fig. 6.4.3b).

a) b)

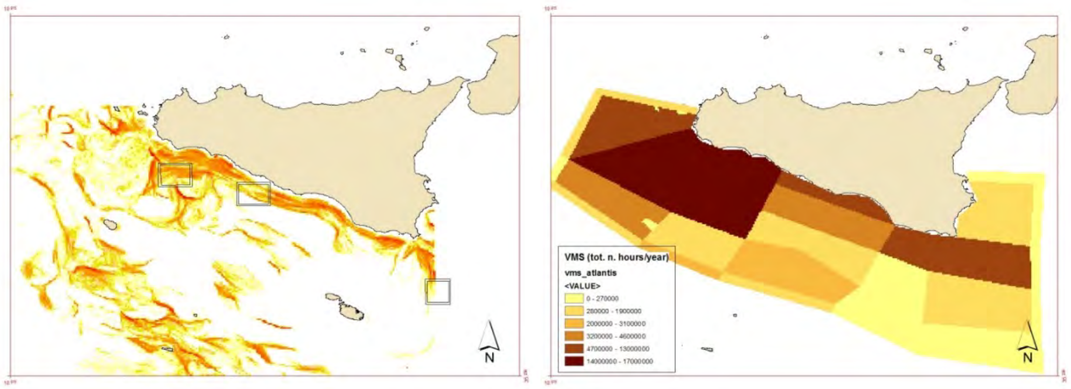

Fig. 6.4.3. Map showing the distribution of fishing pressure in the SoS area and in the established FRAs, calculated as total number of trawl hours in 2015 (a), and in the ATLANTIS boxes (b).

Step 2: Calculate the catch distribution of of HKE and DPS age groups the model domain

We used the annual map of spatial distribution of DPS and HKE juveniles (0 group), obtained in the EU project MEDISEH (Colloca et al., 2015), to calculate the proportion of juveniles included in the FRAs as well as in ATLANTIS boxes (Fig. 6.4.4). FRAs included 26.4% of hake and 19.1% of DPS juveniles, thus showing the potential usefulness of their implementation to consistently reduce mortality on the 0 group.

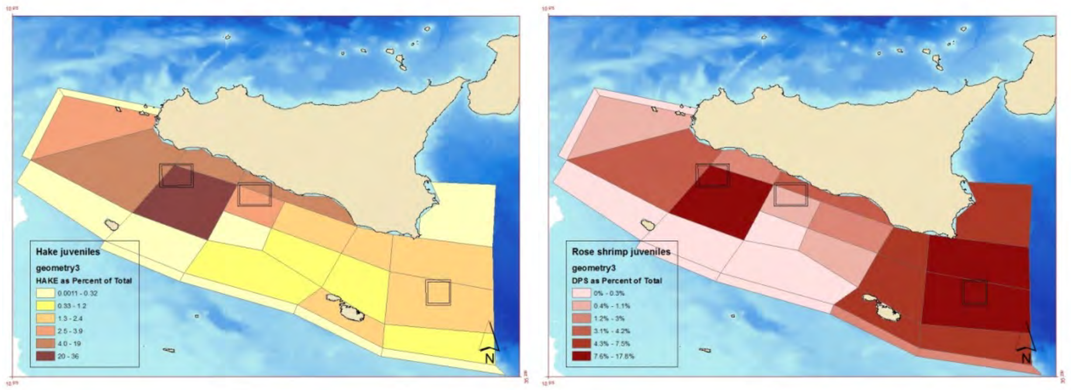

Fig. 6.4.4. Proportional distribution of hake and rose shrimps juveniles in the ATLANTIS model domain. Data are modelled spatial distribution of recruits from MEDITS data 2003-2010 (Colloca et al., 2015).

The overall fisheries catch of the 0 group of the two stocks in 2014 was then splitted in the different ATLANTIS boxes and the associated fishing mortality calculated simply as proportion of the observed total catch:

$C_t=C_1+C_2+C_3+\ldots$

$F_t=F_t*c_1/C_t + F_t*c_2/C_t + F_t*c_3/C_t + \ldots$

Where C_t is the total catch, c_1, c_2, c_3 are the catches in ATLANTIS boxes 1, 2,3 ; F_t is the total fishing mortality at age 0.

For the other age groups n estimated distribution of commercial catches of ages from 1+ to 6+ for HKE and 1+ to 3+ for DPS have been derived combining VMS data with MEDITS data. MEDITS cpue (n km^{-2} by age group) were used to calculate the proportion of each age class in the different ATLANTIS boxes and the overall catch /fishing mortality in the area splitted accordingly.

Step 3: include a spillover effect

The observed proportion of age 0 fish in the FRAs is assumed to widespread as age 1 fish in the adjacent ATLANTIS boxes. The number of age 1 fish deriving from FRA is calculated taking into account the reduction due to natural mortality.

Gear selectivity scenarios

Scenarios will be based on the results of experimental trwal fishing trials carried out in 2016 in the frame of the MINOUW project. Ad-hoc sorting grids were mounted on standard commercial trawl net of an Italian trawler and several hauls were carried out to assess the reliability in reducing the catch of juveniles (0 group specimens, see Annex 6.2). Simulations will be done in 2017 with both ATLANTIS and GADGET and provided for DST prototype II assuming three different proportion of vessels mounting sorting grids on the nets (50, 75, 100%).

Trade marks and ecolobelling

A simulation of the effect of introducing trade marks (i.e. chain of custody), and ecolabelling will be introduced in each management scenarios to assess the benefits of a market strategy based on an increased quality of the products. Trade marks: in Italy, the " Ministero delle Politiche Agricole Alimentari e Forestali" (MIPAAF) is entitled to provide a Protected Designation of Origin (D.O.P.) or Protected Geographical Indication (I.G.T.) certification. The costs are low (i.e. about 1000.00 euro).

Ecolabelling: Friend of the Sea certification. The cost is about 2000 euro per boat or 3.500,00 euro per three boats. Larger fishery vessel society should to pay about 20.000,00 euro for the certification.

Regulate access to fishing grounds

The effect of adopting a list of authorized fishing vessels to fish in the SoS area will be contrasted with the current situation (BAU) where access to fishing grounds in international waters is permitted to all the vessels registered by GFCM. The effects of IUU fishing will be simulated assuming different levels of uncompliance to the measures enforced (i.e. FRAs, reduced numbers of allowed days at sea, etc.).

Climate change scenarios

A simple climate change scenario was set-up to test the ecosystem reaction to a progressive increase in water temperature. Three different scenarios were considered corresponding to increments in the average local sea water temperatures (SWT) over the next 50 years of 1, 2 and 3°C with respect to the present day values.

The spatial and temporal variability of the SWT was reproduced by post-processing of the already existing set of oceanographic data obtained from scenarios IPCC RA and A2. In particular, we estimated, for each box and levels of the model domain, the differences between the temperature values obtained by A2 and RA scenarios on daily basis. This procedure allowed us to estimate a delta matrix systems (DM herefater) describing both the temporal and spatial variability of the variation of T between the two scenarios.

The obtained differences were then averaged obtaining the average difference in water temperature between the RA and A2 scenario at basin scale and for the whole decade.

This value was then compared with the predefined set of new sea water temperature differences, 1, 2 and 3°C respectively, obtaining 3 different multiplication factors. Finally, each multiplication factor was applied to the matrix system DM in order to obtain a new set of water temperature delta values to be added to the RA water temperature dataset. The described procedure allowed to obtain 3 different dataset of temperature forcing, one for each new scenario.

The original plan was to set up a series of scenarios based on the SRES approach taken by the IPCC since 2000, incorporating socio-political drivers as well as projected climate change with two distinct socio-political scenarios, broadly consistent with the IPCC A2 (National Enterprise) and B1 (Global Community) storylines, formulated as part of the FP7 project VECTORS, will be possibly tested. For the future scenarios, values for nutrient levels can be adjusted according to the respective future scenario based on the assessments given in the European Lifestyles and Marine Ecosystems report (www.elme-eu.org, Langmead et al. 2007), where a large change was taken to be 60%, a standard change 30%.

However climate forcing was not identified by local stakeholders as a priority for the management of fisheries in the SoS region. This implies to focus our modelling effort on those management aspects considered relevant for the achievement of the identified management objectives and the climate scenarios will not be produced for DST II. The possibility to explore climate scenarios will depend also by the availability of time for the development of the scenarios required for the DST II.

Results from the model

Business as usual (BAU)

ATLANTIS

Results from the model simulation, under the BAU scenario, show a progressive decrease in hake biomass and landings (Fig. 6.5.1) in the period of constant fishing mortality (Fig. 6.4.1), before 2008 and after 2014. Hake is currently overexploited (Fig. 6.5.8) and its biomass and landings are highly sensitive to changes in fishing mortality. The model output for biomass and catches and fishing mortality are comparable in magnitude and trend to the estimates done using the Extended Survivors Analysis (XSA, Fig. 6.5.1-6.5.2).

The forecasted spatial distribution of hake in 2020 and 2030 is shown in Fig. 6.5.3 During this period the abundance of hake increases in the Eastern area of the SoS.

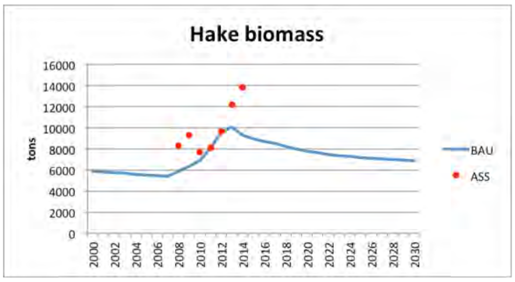

Fig. 6.5.1. Hake biomass and landings under the BAU scenario. Dots represent assessment estimates.

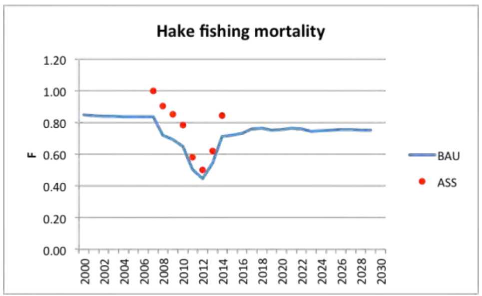

Fig. 6.5.2. Hake fishing mortality under the BAU scenarios. Dots represent estimates from stock assessment (XSA).

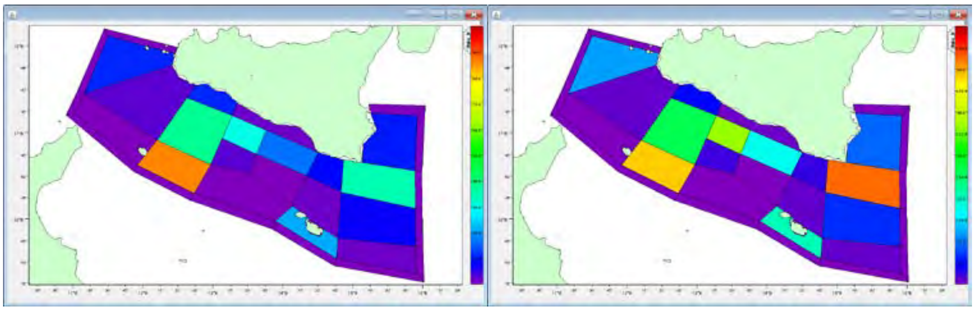

Fig. 6.5.3. Hake spatial distribution in 2020 (sx) and 2030 (dx)

The model reproduced the biomass and landing of deep water rose shrimp within range of assessment estimate although underestimating the inter-annual fluctuations (Fig. 6.5.4). Since DPS is currently modelled using the Atlantis format for invertebrates (biomass pool and not numbers) it was not possible to extract fishing mortality. The future plan is to restructure DPS using the vertebrate framework of Atlantis (different age classes, size, numbers at sea, etc.). The forecasted spatial distribution of total DPS biomass in 2020 and 2030 is shown in Fig. 6.5.5. The model reproduces the

DPS distribution within the SoS with highest biomass in the eastern part of the Strait. The biomass in the area of the Adventure Bank area is lower than observed and distributed offshore.

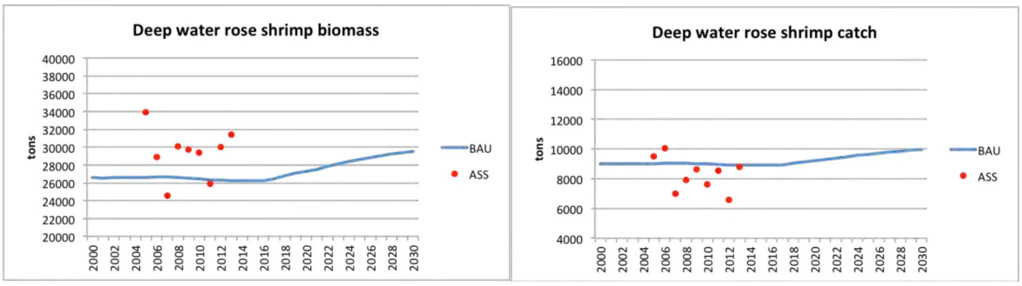

Fig. 6.5.4. Deep water rose shrimp biomass and landings under the BAU scenario. Dots represent assessment estimates.

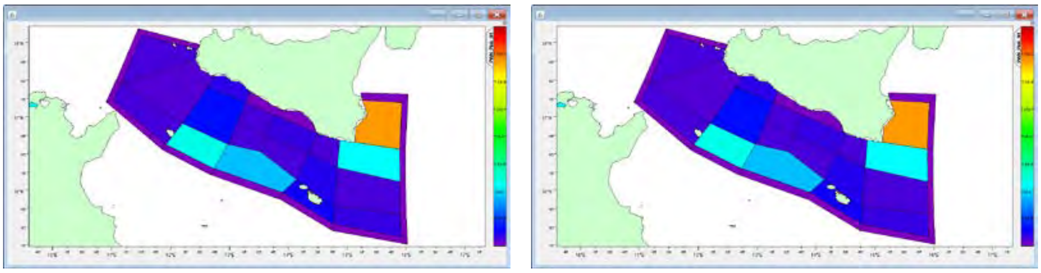

Fig. 6.5.5. Deep water rose shrimp spatial distribution in 2020 (sx) and 2030 (dx)

The biomass of all other target species estimated by the model is within range and with the same trends of assessment estimates in Fig. 6.5.6.

The traffic light plot (Fig. 6.5.7) shows the evolution of the whole ecosystem biomass. A positive relative change in biomass since 2000 is shown in green, while a negative change is shown in red.

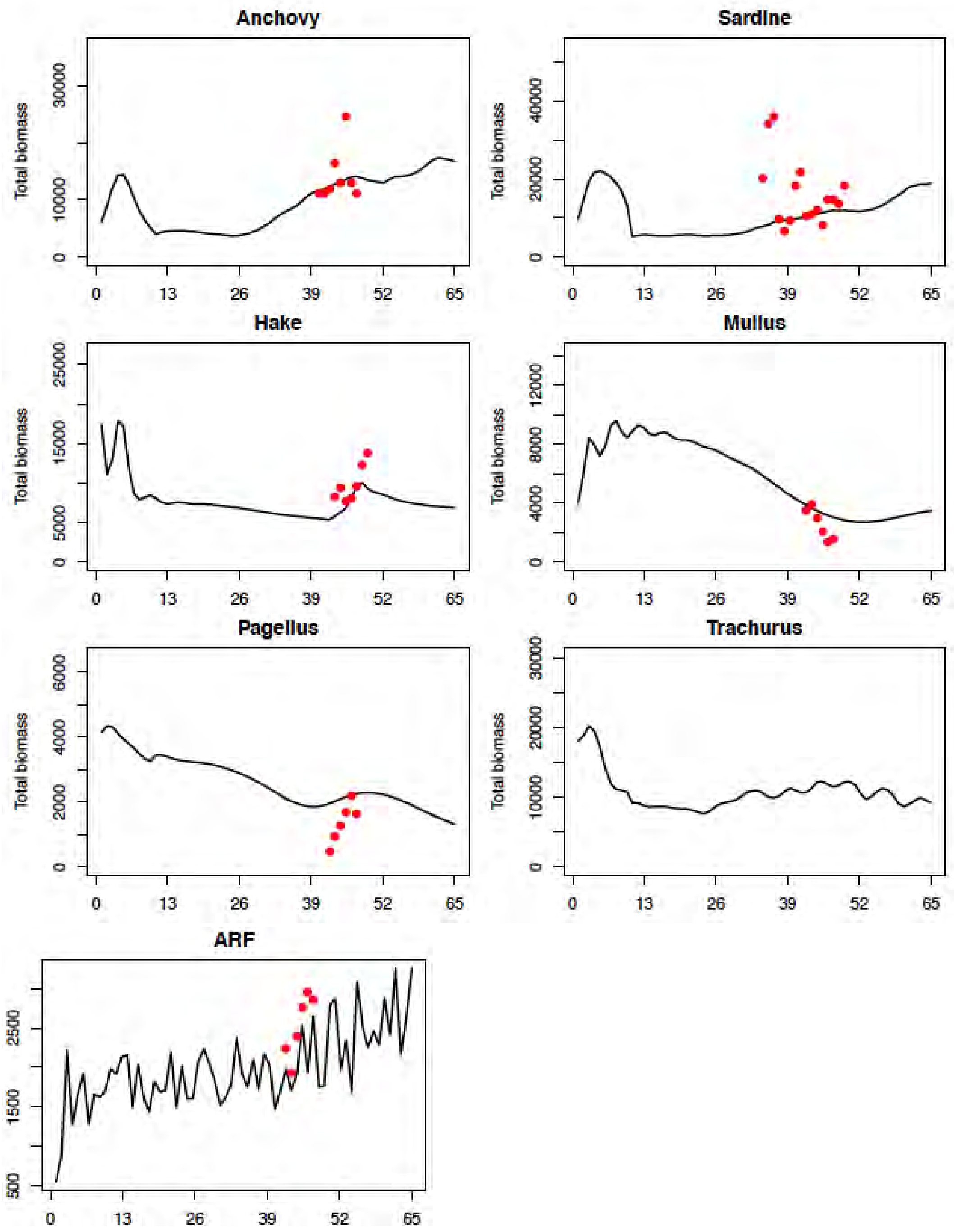

Fig. 6.5.6. Other target species biomass under BAU scenario. Year 0 :1965; year 50: 2015; year 55: 2020; year 65: 2030. Anchovy (Engraulis encrasicolus), sardine (Sardina pilchardus), hake, red mullet (Mullus barbatus), red pandora (Pagellus erythrinus), giant red shrimp (Aristeomorpha foliacea).

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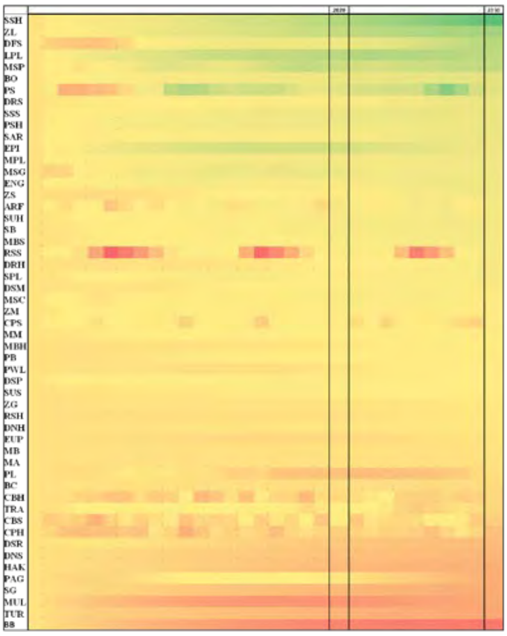

Fig. 6.5.7. Traffic light plot of change in biomass of all groups relative to year 2000. Green shows a positive change; red a negative change (code as in Table 6.3.1.).

GADGET

The current multispecies GADGET model was adapted to simulate a BAU scenario where it is assumed a constant catch of the three stocks until 2025 (Fig. 6.5.8). A reduction trend in HKE and HOM biomass is predicted, whereas DPS does not show major changes.

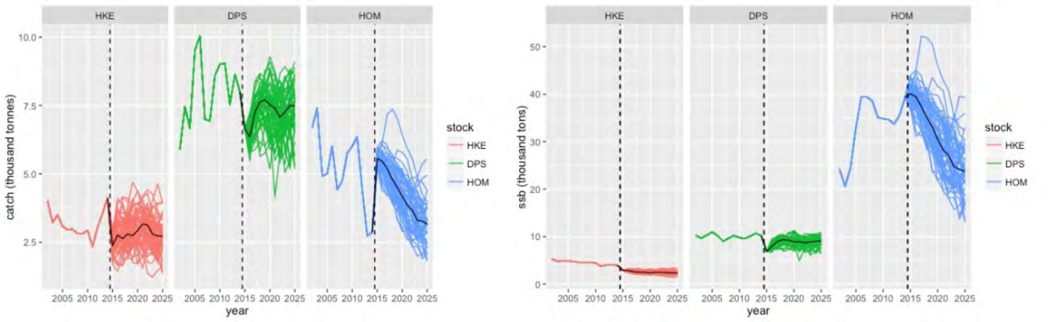

Fig. 6.5.8. Multispecies GADGET model. Stochastic simulations of a BAU scenario for fisheries catch and total biomass of hake, deep water rose shrimp and horse mackerel.

F_{msy} estimation for Hake and Deep water rose shrimp

A series of simulations were run to estimate the F level that optimizes catches (F_{MSY}) for both HKE and DPS. The BAU simulation was used a baseline and run with increasing F values for HKE and DPS separately and the estimated catches were used to produce yield curves (Fig. 6.5.9). The F_{MSY} for HKE was estimated to be 0.2 (green arrow), which is close to F_{01} estimate obtained with XSA (FLR BRP library) of 0.18 (red arrow). The model estimate of F_{MSY} for DPS is 1.1 that is slightly above the current level of fishing mortality (1.0) and the XSA assessment estimate of F_{MSY} that is 0.93.

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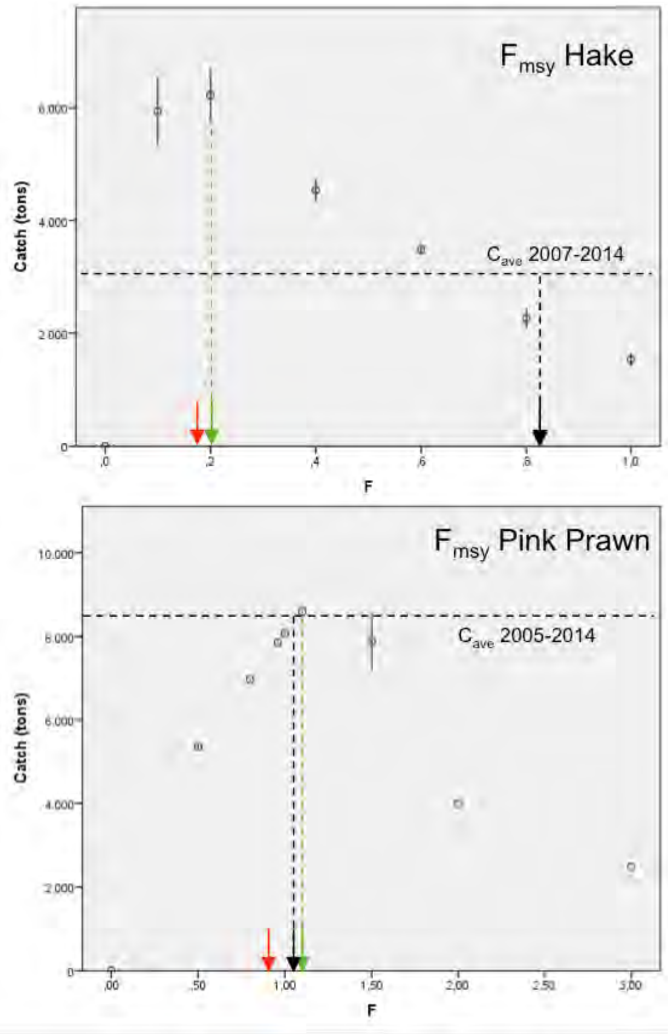

Fig. 6.5.9. Model reproduction of the yield curves for hake and deep water rose shrimp. The red arrow points at FMSY estimated by the assessment; The black arrow shows the current F; the green arrow shows the FMSY estimated by Atlantis. The horizontal dotted line show the average catch for the available periods.

F reduction scenarios

It implies a gradual reduction of fishing mortality from current F (Fc_{urr}) to F_{MSY} (Table 6.4.3). The effect of introducing a management rule that reduces fishing mortality to F_{MSY} level by 2020 is investigated in three fishing reduction scenarios:

5% annual reduction corresponds to the change from F_{curr} to Fmsy for DPS,

20% annual reduction corresponds to the change from F_{curr} to Fmsy for HKE,

15% is an intermediate level of F reduction.

Fig. 6.5.10 shows the forecasted effects of F reduction on hake biomass and catch. It can be seen that immediately after the implementation of the F reduction management rule (2015) the biomass starts to increase. The rate of biomass increase was directly proportional to the level of fishing mortality

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reduction for all scenarios. At the same time, catches decrease, with fishing mortality reduction, until 2020, after which year F is kept constant to the 2020 value (Fig. 6.5.11), and catches increase steadily until the end of the simulated period (2030). The trend in catch rise after 2020 approximates a plateau at the 2030 value for the 5% reduction scenario, while for the 15% and 20% scenarios the trend in catches after year 2020, suggest that caches would keep on increasing after 2030.

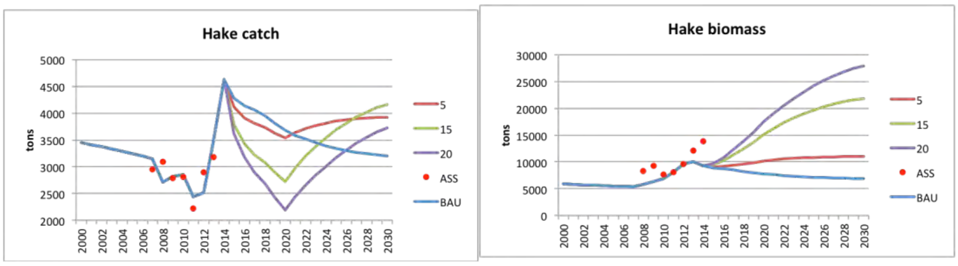

Fig. 6.5.10. Hake biomass and landings under the effort reduction scenarios. Dots represent assessment estimates.

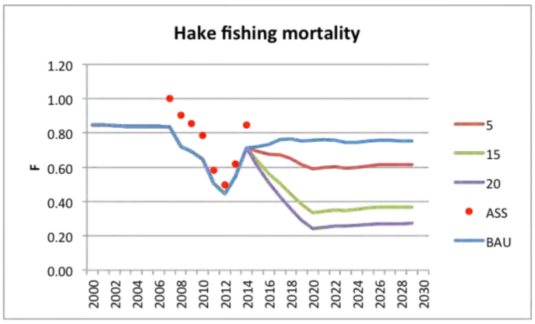

Fig. 6.5.11. Hake fishing mortality under the effort reduction scenarios. Dots represent stock assessment estimates (XSA).

Similar trend was predicted also for DPS although with a much weaker response to changes in F (Fig. 6.5.12). DPS biomass increased as the F reduction is implemented and at the same time, catches reduce. Catches start to increase when F, after 2020, is kept at a constant level.

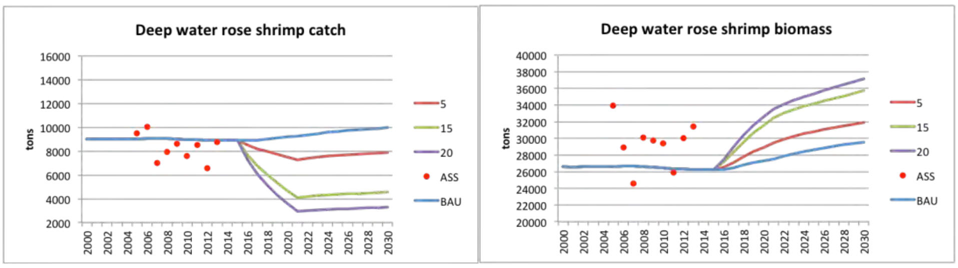

Fig. 6.5.12. Deep water rose shrimp biomass and landings under the effort reduction scenarios. Dots represent stock assessment estimates (XSA).

Table 6.5.1 illustrates the effect of the F reduction management scenarios on the biomass and catch for HKE and DPS in year 2020 and 2030. It also shows the percentage change compared to the 2020 and 2030 values obtained in the BAU scenario.

It can be seen that while for HKE the catch reduction in 2020 will be compensated by increased biomass at sea and catches in 2030 for all scenarios, for DPS which is currently being fished closed, or for the model, slightly below F_{MSY} (Fig. 6.5.10) decreases in F results in reduced catches both in the short-term (2020) and in the medium-long term (2030).

Table 6.5.1. Predicted biomass and catch for HKE and DPS in year 2020 and 2030 as absolute values in tons and percentage change from the BAU projection.

	BIOMASS HKE				**CATCH HKE**			
tons	**BAU**	**5**	**15**	**20**	**BAU**	**5**	**15**	**20**
2020	7772	10131	15097	17460	3679	3535	2717	2192
2030	6885	10988	21861	27961	3200	3924	4166	3730
% change								
2020	1	30.3	94.2	124.6	1	-3.9	-26.1	-40.4
2030	1	59.6	217.5	306.1	1	22.6	30.2	16.6

	BIOMASS DPS				**CATCH DPS**			
tons	**BAU**	**5**	**15**	**20**	**BAU**	**5**	**15**	**20**
2020	27288	28927	31568	32583	9209	7514	4681	3572
2030	29555	31880	35708	37145	9967	7912	4552	3292
% change								
2020	1	6.0	15.7	19.4	1	-18.4	-49.1	-61.2
2030	1	7.9	20.8	25.7	1	-20.6	-54.3	-67.0

Finally, an economic simulation was carried out on the effects of reduction in number of fishing days on the gross profits generated by the HKE and DPS landings of the Italian trawlers (Fig. 6.5.11b). The data for the simulations are reported in ANNEX 6.2. The costs associated at the fishing days at sea have been obtained from linear regressions between type of costs and days at sea for the period 2004-

2015. DPS made 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. 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. To simulate the effects of reducing fishing days on gross profits, costs were considered as dependent by the number of days at sea only. In addition, the prize 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 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 case of DPS all the scenarios predict a reduction in the average daily profits compared with the business as usual (BAU) scenario. The DPS stock appears to be exploited very close to F_{MSY} and therefore any major reduction in F returns a predicted decreasing in landings and associated revenues and profits. In the case of HKE the current exploitation will produce a decreasing in profits, whereas the -15% and -20% fishing days scenarios rebuild quickly profits after 4 years of steady decline. The 5% reduction results useful to keep almost constant the profits in the medium term.

Hake **Deep water rose shrimp**

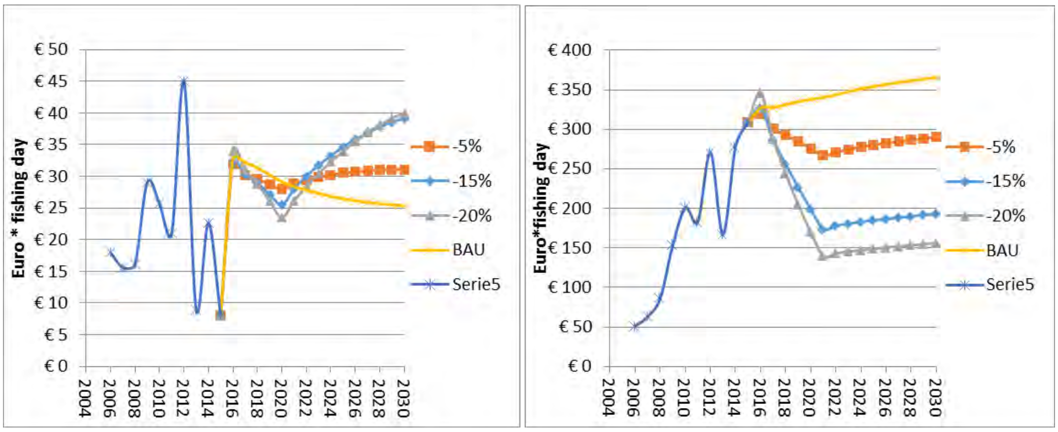

Fig. 6.5.11b. Average profits of Italian trawl fleet, as euro per fishing days, generated by the landings of hake and deep-water rose shrimp in the SoS under different fishing effort reduction.

GADGET

A set of preliminary simulations on the effect of a reduction of the harvestable biomass was done based on a GADGET multispecies model assuming annual recruitment for HKE and DPS. Simulations show the effects of the trade-offs due to the prey-predator relationships. The rebuilding of HKE stock due a reduction in the catch of the exploitable biomass bring to a strong reduction in the biomass and catch of its main prey (HOM). The impact on the DPS stock change consistently moving from a 20% to a 50% reduction. In the first case the stock biomass returns to the historical levels whilst in the 50% reduction the quick rebuilduing of the HKE stock bring DPS biomass well below the historical levels.

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20% reduction in the exploitable biomass

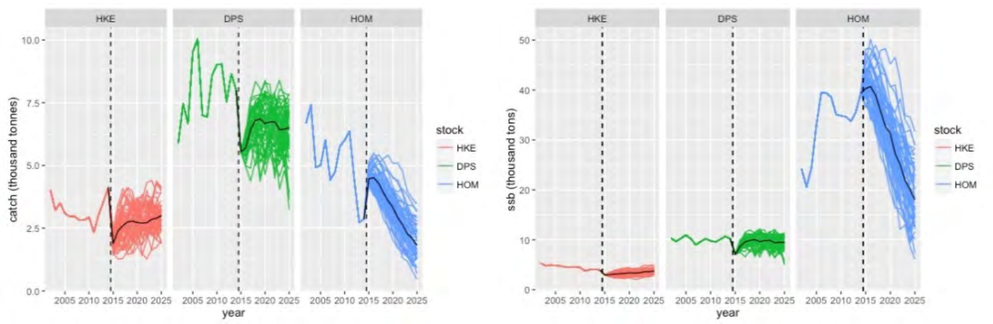

Fig. 6.5.12. Multispecies GADGET model. Trend catch and total biomass of hake, deep-water rose shrimp and horse mackerel assumin a reduction of 50% in the catch of the harvestable biomass until 2025.

50% reduction in the exploitable biomass

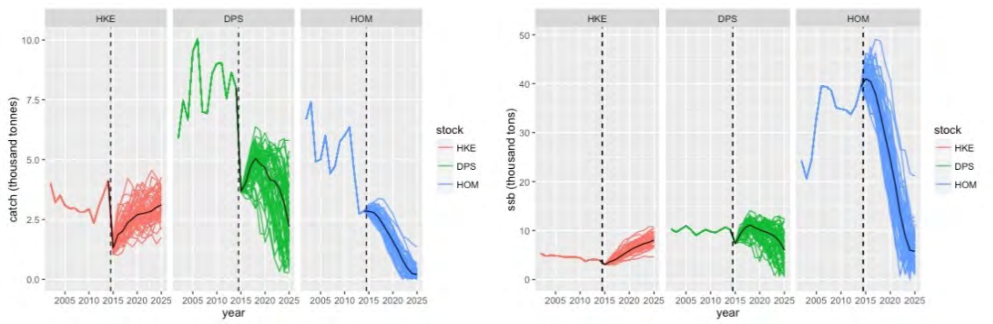

Fig. 6.5.13. Multispecies GADGET model. Trend catch and total biomass of hake, deep-water rose shrimp and horse mackerel assumin a reduction of 50% in the catch of the harvestable biomass until 2025.

Fisheries Restricted Areas

A preliminary simulation of the effect of the institution of the threes FRAs in the Strait of Sicily on HKE and DPS biomass and catch is shown in Figs. 6.5.14-6.5.16. This has a positive effect on the biomass at sea of hake (Fig. 6.5.12), resulting in an increase from the BAU scenario of 5% in 2020 and over 30% in 2030, as result of decreased fishing mortality on juveniles (Fig. 6.5.15).

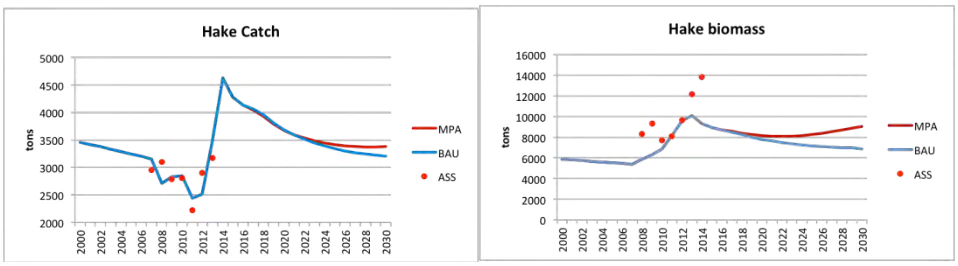

Fig. 6.5.14. Hake biomass and landings under the FRA scenario. Dots represent stock assessment estimates (XSA).

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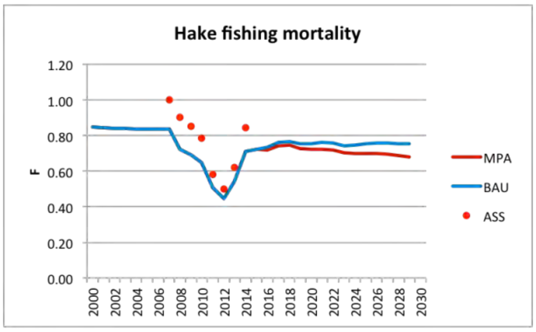

Fig. 6.5.15. Hake F mortality under the FRA scenario. Dots represent stock assessment estimates (XSA).

The effect on DPS is negligible, with a slight increase in biomass, but a decrease in catches. At this stage, more work is needed to improve the fishing effort reallocation and to clarify the mechanisms underlying the simulated trends, in particular the trophic dynamics within the ecosystem.

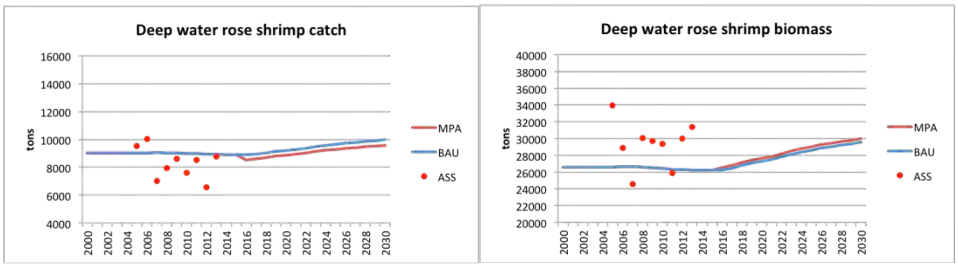

Fig. 6.5.16. Deep water rose shrimp biomass and landings under the FRA scenario. Dots represent stock assessment estimates (XSA).

Results of area closure simulations are showed in Table 6.5.2.

Table 6.5.2. Predicted biomass and catch for HKE and DPS in year 2020 and 2030 as absolute values in tons and percentage change from the BAU projection.

	BIOM HAKE		CATCH HAKE		BIOM DPS		CATCH DPS	
	BAU	MPA	BAU	MPA	BAU	MPA	BAU	MPA
2020	7772	8139	3679	3670	27288	27633	9209	8871
2030	6885	9011	3200	3381	29555	29967	9967	9577
% change								
2020	1	4.7	1	-0.3	1	1.3	1	-3.7
2030	1	30.9	1	5.7	1	1.4	1	-3.9

A more complex and reliable scenarios will be implemented for DST II.

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Climate scenario

This preliminary set of simulations investigate the effect of gradual increase of water temperature on the ecosystem structure, in particular HKE and DPS. These results have to be considered as a proof of test of the system, as a prediction of the effect on temperature rise would require an accurate exploration of the physiological effect from laboratory experiments and literature. For running these simulation the default setting of Atlantis have been left untouched. Fig. 6.5.17 shows for all the scenario tested the negative effect of temperature rise on hake biomass and catches, leading to a 10% reduction of catches in the long-term (2030) under the 3°C scenario.

For all scenario DPS biomass and catches are considerably increased under the climate change scenario (Fig. 6.5.18). Further investigation is needed to tackle this scenario.

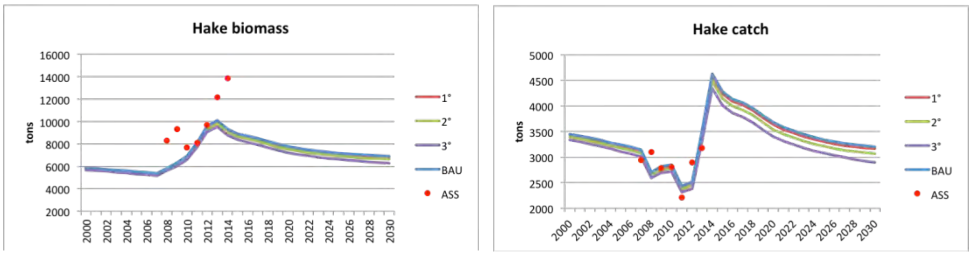

Fig. 6.5.17 Hake biomass and landings under the warming climate scenario. Dots represent stock assessment estimates (XSA).

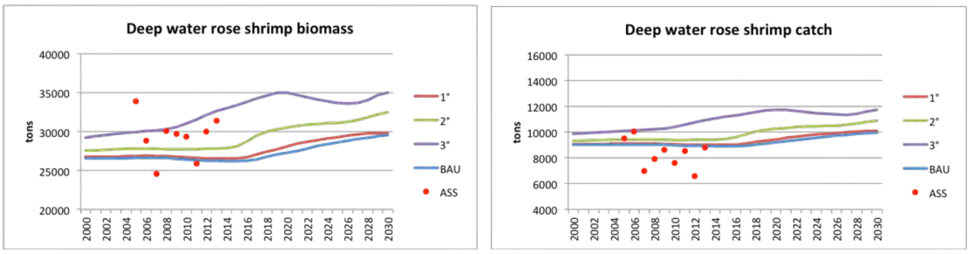

Fig. 6.5.18 Deep water rose shrimp biomass and landings under the warming climate scenario. Dots represent stock assessment estimates (XSA).

A summary of the simulations of the effects of warming on HKE and DPS stocks are shown in Table 6.5.3.

Table 6.5.3. Predicted biomass and catch for HKE and DPS in year 2020 and 2030 as absolute values in tons and percentage change from the BAU projection

	BIOMASS HAKE				**CATCH HAKE**			
tons	**BAU**	**1**	**2**	**3**	**BAU**	**1**	**2**	**3**
2020	7772	7685	7496	7228	3679	3634	3539	3400
2030	6885	6812	6627	6277	3200	3163	4166	3730
% change								
2020	1	-1.1	-3.6	-7.0	1	-1.2	-3.8	-7.6
2030	1	-1.1	-3.7	-8.8	1	-1.2	-4.1	-9.6

	BIOM DPS				**CATCH DPS**			
tons	**BAU**	**1**	**2**	**3**	**BAU**	**1**	**2**	**3**
2020	27288	28114	30551	35013	9209	9450	10266	11757
2030	29555	29883	32531	34996	9967	10091	10903	11750
% change								
2020	1	3.0	12.0	28.3	1	2.6	11.5	27.7
2030	1	1.1	10.1	18.4	1	1.2	9.4	17.9

Linking the ecosystem model output with stakeholder request

Ecosystem models, Atlantis and GADGET, link with the stakeholder interest through the decision support tool. It uses output from the ecosystem models as the science based knowledge as estimates about the expected consequences under alternative management choices. The MCA value tree articulates the variables that stakeholders have identified as pivotally important for them (Fig. 6.6.1).

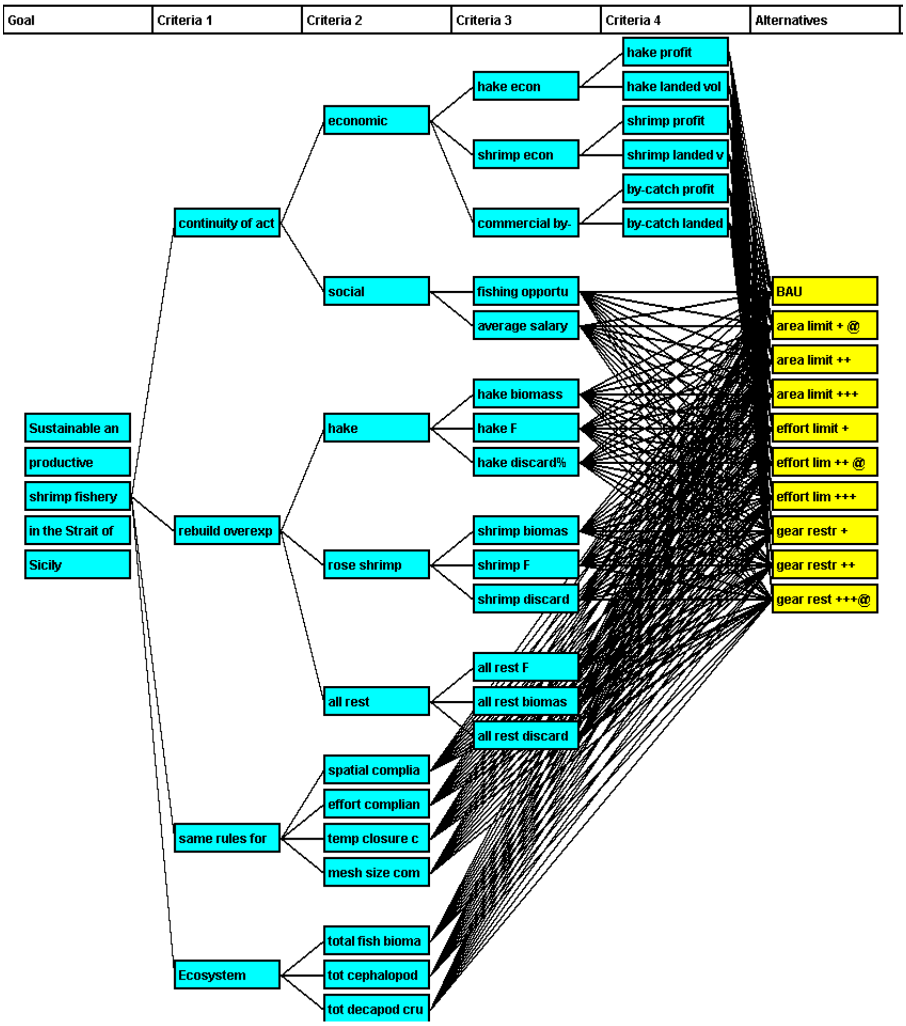

Fig. 6.6.1. Value tree for the SoS trawl fisheries.

The ATLANTIS model is currently capable of simulating future projections for most of the variables identified in the value tree. These include landed volume, revenues, costs and profits for shrimp and hake. Estimates of these variables for the commercial by-catch species would require a further implementation of the current Atlantis model. Also biomass and fishing mortality rate can be simulated forward under the alternative management scenarios. Fishing mortality rate will be used as proxy for fishing opportunities (i.e. days at sea), albeit the relationship between these two variable is still not very well understood. Currently there is a plan on how to estimate average salary of the crew which is conditioned by the amount of landings in relation to the type of work contract applied (i.e. “alla parte contract”). ATLANTIS can, in its current state, project discard rate of hake 0-age group. It is not yet possible for rose shrimp due because, as invertebrate, the species is simulated as a biomass pool . Also, an approach to estimate the four compliance indicators is yet to be developed.

Summary of best ATLANTIS models outputs

The overall effects in the short and of the different scenarios for target and major groups are summarized in Table 6.7.1.

Fig. 6.7.1 shows that for HKE F reduction and FRA measures lead to a long-term increase both in biomass and catch. For DPS the fishing mortality reduction scenario results in an increase in biomass but a decrease in catch, which is a reasonable consequence considering that DPS fishing mortality is very close to F_{MSY}.

The effect on water temperature increase generates a general increase in biomass of the pelagic community and giant red shrimp, while it has a negative effect on demersal fish biomass (Fig. 6.7.2).

Table. 6.7.1 - Percentage changes from BAU estimates for all target species for 2020 and 2030. Green indicates a positive change; red negative change.

2020	F 5%	F 15%	F 20%	FRA	+1°C	+2°C	+3°C
HAK	30.34	94.24	124.64	4.72	-1.12	-3.55	-7.00
DRS	6.00	15.68	19.40	1.26	3.02	11.96	28.31
ENG	0.00	0.00	0.00	-4.93	3.13	44.19	55.17
SAR	0.00	0.00	0.00	1.76	-2.14	-2.90	-5.39
MUL	-0.02	-0.06	-0.08	-3.53	-0.18	-0.87	-3.39
PAG	-0.01	-0.02	-0.03	10.44	-0.19	-0.62	-3.91
TRA	-0.05	-0.15	-0.19	13.66	0.76	9.53	4.59
ARF	-0.02	-0.07	-0.09	6.72	13.07	51.97	127.12
PEL	-0.01	-0.01	-0.01	0.19	2.64	104.69	158.56
CEP	0.00	-0.01	-0.01	0.74	2.31	7.02	16.29
ALL	0.00	0.00	0.00	0.00	0.00	-0.03	-0.05

2030	F 5%	F 15%	F 20%	FRA	+1°C	+2°C	+3°C
HAK	59.60	217.53	306.13	30.88	-1.06	-3.75	-8.83
DRS	7.87	20.82	25.68	1.39	1.11	10.07	18.41
ENG	0.01	0.00	-0.01	-6.71	2.45	75.89	89.11
SAR	0.01	0.03	0.07	12.39	0.40	4.83	8.91
MUL	-0.06	-0.19	-0.24	62.55	0.16	-1.62	-2.02
PAG	-0.05	-0.14	-0.19	50.51	-1.66	-5.81	-13.38
TRA	-0.26	-0.79	-1.02	93.06	5.91	28.80	23.38
ARF	-0.10	-0.27	-0.34	3.55	14.37	56.44	155.45
PEL	-0.02	-0.11	-0.16	5.23	3.14	199.39	217.98
CEP	-0.01	-0.03	-0.03	0.46	2.29	5.07	12.96
ALL	0.00	0.00	0.00	0.00	0.00	-0.06	-0.07

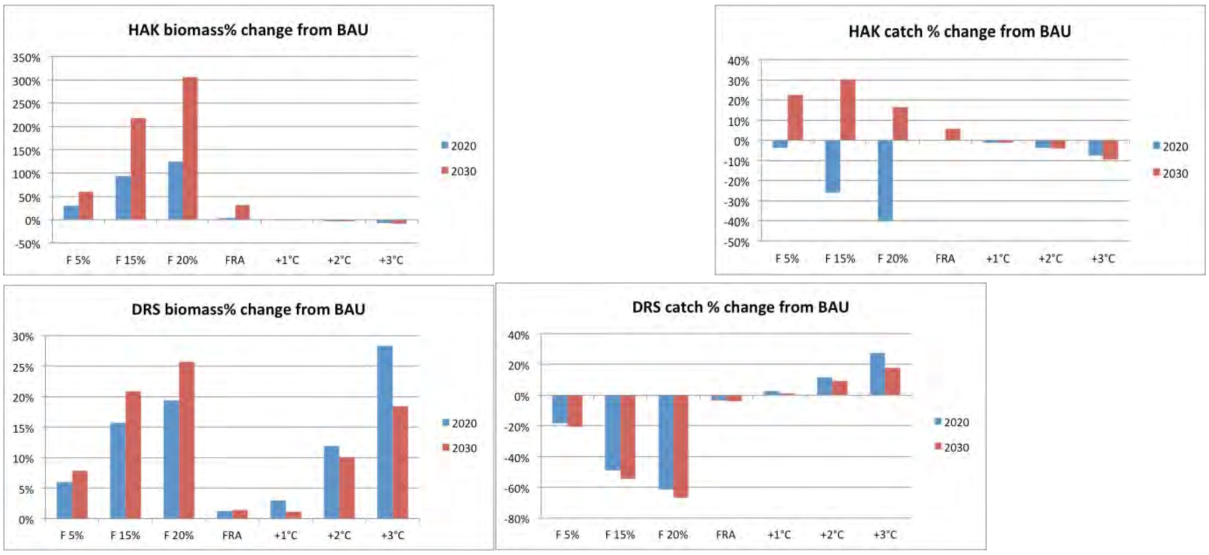

Fig. 6.7.1. The effect of different scenarios on biomass and catch of HKE and DPS in 2020 and 2030 compared to the BAU scenario.

A general outlook at the results of the different scenarios on the other major groups reveals that the FRA scenario estimates a positive effect on the biomass of most species, in particular those abundant in the FRA area, such as MUL and PAG. The water temperature increasing scenarios, in particular the extreme 2° and 3°, predict an ecosystem response in favour of short-living fast growing species, such as CEP and PEL, which may benefit of the decreased competition for food by slower growing fish species. However, more analysis on these scenarios is needed to assess the mechanisms of this

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response by the ecosystem. The effect of the fishing mortality reduction for HKE and DPS does not have a significant effect on the other species.

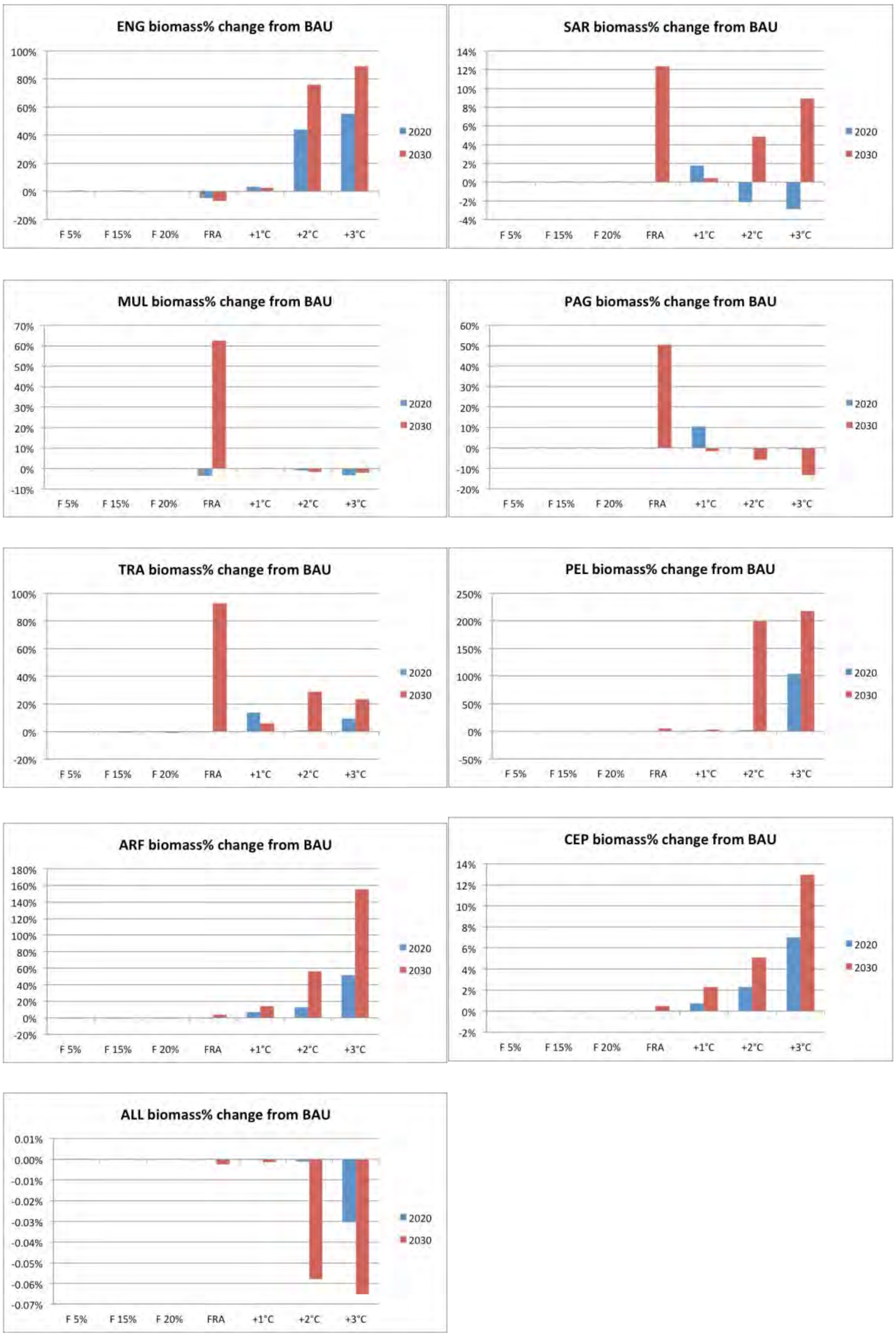

Fig. 6.7.2. The effect of different scenarios on biomass of target and major groups in 2020 and 2030 compared to the BAU scenario.

Scenarios implementation

In 2017 the scenarios developed so far with ATLANTIS will be improved and refined by both increase their complexity adding new process sub-models in the scenarios framework. In particular the following aspects will be evaluated for their inclusion:

I. Include migration pattern of juveniles ok key stocks (i.e. HKE and DPS) to model the spillover effect generated by area clusures;
II. Model DPS using the ATLANTIS framework used for vertebrates (division in multiple age-classes, and explicit individual weight and size).
III. Focus on the change in species and size composition of catches per fleet.
IV. Model the re-allocation of fishing effort as effect of area closure measures (i.e. FRAs) using VMS data.
V. Account for the relationship between fishing effort (i.e. days at sea, nominal effort) and fishing mortality.
VI. Estimate the set of ecological indicators identified as suitable to detect the effect of management measure on the ecosystem structure and functioning. Indicators must be quantifiable using ATLANTIS simulation outputs and sensible to changes in fishing mortality/catch.
VII. Temperature effects on key stocks (i.e recruitment) and if possible ecosystem processes (i.e. primary production).

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7. Black Sea case study

Brief description of the case study objectives

The GADGET model developed included 7 different populations or functional groups (turbot, anchovy, sprat, gobies, whiting, molluscs, cetaceans) with 3 fleets acting in 3 different areas (Romanian area, West Black Sea area and all Black Sea). Both the impact of the interactions between species and the impact of fisheries harvesting the species have been included in the model.

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/km2/year), IUU catches (t/km2/year), P/B= Z (total mortality), Q/B (consumption rate). Turbot diet novel data obtained from a stomach content analysis it is first implemented in a multispecies model in the area.

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. G.1). 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 . So, generally, the favourite prey are gobies.

Modelling framework

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 (Fig 7.1).

	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
1	cetaceans																							
2	turbot1	0.0010000																						
3	turbot2	0.0050000																						
4	turbot3	0.015000																						
5	turbot4	0.015000																						
6	turbot5	0.025000																						
7	turbot6	0.025000																						
8	turbot7																							
9	turbot8																							
10	turbot9																							
11	anchovy1								0.020000	0.020000														
12	anchovy2	0.053000							0.020000	0.020000														
13	anchovy3	0.053000				0.0100000			0.020000	0.020000														
14	anchovy4	0.053000				0.020000			0.020000	0.020000														
15	anchovy5	0.040000				0.020000			0.020000	0.020000														
16	sprat1	0.010000				0.040000	0.030000	0.020000	0.060000	0.060000	0.070000													
17	sprat2	0.010000				0.040000	0.030000	0.020000	0.060000	0.060000	0.070000													
18	sprat3	0.010000				0.040000	0.030000	0.020000	0.060000	0.060000	0.070000													
19	sprat4	0.010000				0.040000	0.030000	0.020000	0.060000	0.060000	0.070000													
20	sprat5	0.010000				0.040000	0.030000	0.020000	0.060000	0.060000	0.070000													
21	whiting1	0.025000				0.040000	0.080000	0.060000	0.040000	0.060000	0.050000													
22	whiting2	0.025000				0.040000	0.080000	0.090000	0.040000	0.050000	0.050000													
23	whiting3	0.025000				0.040000	0.080000	0.080000	0.030000	0.050000	0.050000													
24	whiting4	0.025000				0.040000	0.080000	0.060000	0.030000	0.050000	0.050000													
25	whiting5	0.015000					0.070000	0.060000	0.030000	0.050000	0.050000													
26	whiting6	0.0050000						0.020000	0.030000	0.040000	0.050000													
27	goby1					0.050000	0.050000	0.070000	0.050000	0.020000	0.050000													
28	goby2	0.015000				0.10000	0.080000	0.12000	0.10000	0.070000	0.080000													
29	goby3	0.015000				0.10000	0.080000	0.12000	0.10000	0.070000	0.080000													
30	goby4	0.015000				0.10000	0.080000	0.14000	0.10000	0.14000	0.14000													
31	Mussels					0.050000	0.050000	0.050000	0.050000															
32	Zoobenthos			0.25000	0.60000	0.19000	0.12000	0.030000															0.15000	0.50000
33	Zooplankton		1.00000	0.75000	0.40000							0.25000	0.58000	0.68000	0.90000	0.97000	0.47000	0.80000	0.86000	0.90000	0.96000	0.75000	0.70000	0.45000
34	Phytoplankton											0.75000	0.42000	0.32000	0.10000	0.030000	0.53000	0.20000	0.14000	0.10000	0.040000	0.25000	0.15000	0.050000
35	Detritus																							
36	Import	0.50000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000
37	Sum	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000	1.00000
38	(1 - Sum)	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000

Fig.7.1 - Screenshot of diet composition (DC) matrix from Ecopath showing the predator -prey interactions for Black Sea model

Scenarios description

For develop our scenarios,we consider three kind of measures:

- Business As Usual = 100 % IUU
- Soft Measures = 50 % IUU
- Hard Measures = NO IUU

And three kind of Harvest Control Rule:

- Fishing Mortality(F)
- Total Allowable Catch(TAC)
- Maximum Sustainable Yield(MSY)

The desire output: SSB, catches, landing.

Dataset used it is from three areas:

- national stock (Romania)
- western part of the Black Sea (Ukraine, Romania, Bulgaria)
- stock unique (for all six riparian countries - Bulgaria, Georgia, Romania, Russian Federation,Turkey and Ukraine)

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Main Model - Ecopath with Ecosim(EwE)

	Group name	Trophic level	Habitat area (fraction)	Biomass in habitat area (t/km²)	Biomass (t/km²)	Z (/year)	Production / biomass (/year)	Consumption / biomass (/year)	Ecotrophic efficiency	Production / consumption
1	cetaceans	4.06386	1.00000	0.060000	0.060000		0.20000	4.10000	0.00000	0.04878
	turbot									
2	turbot1	3.00000	1.00000	0.0012355	0.0012355	0.55230		4.05013	0.36050	0.13637
3	turbot2	3.08000	1.00000	0.0099090	0.0099090	0.31220		2.08293	0.73948	0.14988
4	turbot3	3.19200	1.00000	0.020328	0.020328	0.43217		1.47754	0.80797	0.29249
5	turbot4	3.88671	1.00000	0.023977	0.023977	0.44026		1.21399	0.95631	0.36265
6	turbot5	3.94920	1.00000	0.022509	0.022509	0.41620		1.07549	0.95128	0.38699
7	turbot6	4.07591	1.00000	0.018930	0.018930	0.40020		0.99447	0.92038	0.40242
8	turbot7	3.98503	1.00000	0.014921	0.014921	0.38880		0.94383	0.04051	0.41194
9	turbot8	4.03437	1.00000	0.011298	0.011298	0.38023		0.91082	0.00000	0.41746
10	turbot9	4.07302	1.00000	0.028704	0.028704	0.37356		0.86000	0.00000	0.43437
	anchovy									
11	anchovy1	2.25000	1.00000	0.42988	0.42988	2.00000		11.1846	0.00099	0.17882
12	anchovy2	2.58000	1.00000	1.05100	1.05100	1.80000		5.66367	0.00858	0.31782
13	anchovy3	2.68000	1.00000	0.57543	0.57543	1.70000		3.91145	0.01578	0.43462
14	anchovy4	2.90000	1.00000	0.19321	0.19321	1.70000		3.20442	0.04371	0.53052
15	anchovy5	2.97000	1.00000	0.13200	0.13200	0.90000		2.69400	0.09302	0.33408
	sprat									
16	sprat1	2.47000	1.00000	0.42292	0.42292	2.90000		13.8709	0.01830	0.20907
17	sprat2	2.80000	1.00000	0.87624	0.87624	1.63000		6.43272	0.02292	0.25339
18	sprat3	2.86000	1.00000	0.57327	0.57327	1.62000		4.39305	0.03760	0.36876
19	sprat4	2.90000	1.00000	0.21928	0.21928	1.61000		3.54444	0.08241	0.45423
20	sprat5	2.96000	1.00000	0.098534	0.098534	1.45000		3.00800	0.16810	0.48205
	whiting									
21	whiting1	2.75000	1.00000	0.021615	0.021615	1.00000		9.51295	0.59746	0.10512
22	whiting2	2.89800	1.00000	0.077040	0.077040	1.52900		5.12981	0.13530	0.29806
23	whiting3	3.11000	1.00000	0.053090	0.053090	1.38200		3.61024	0.19282	0.38280
24	whiting4	3.21590	1.00000	0.022480	0.022480	1.48700		3.00662	0.38083	0.49457
25	whiting5	3.28836	1.00000	0.0074046	0.0074046	1.34700		2.69749	0.87080	0.49935
26	whiting6	3.35113	1.00000	0.0033047	0.0033047	1.34700		2.49000	0.82556	0.54096
	gobies									
27	goby1	3.14800	1.00000	0.0058134	0.0058134	1.30000		6.48785	0.88917	0.20037
28	goby2	3.21840	1.00000	0.029857	0.029857	0.80000		3.24055	0.66285	0.24687
29	goby3	3.37160	1.00000	0.044180	0.044180	0.70000		2.24873	0.52078	0.31129
30	goby4	3.41750	1.00000	0.11573	0.11573	0.70000		1.61000	0.21766	0.43478
31	Mussels	2.10000	1.00000	5.50000	5.50000		2.97000	8.00000	0.01522	0.37125
32	Zoobenthos	2.32000	1.00000	0.25545	0.25545		2.50000	3.50000	0.77919	0.71429
33	Zooplankton	2.00000	1.00000	9.30000	9.30000		30.0000	60.0000	0.08104	0.50000
34	Phytoplankton	1.00000	1.00000	4.39000	4.39000		135.000		0.99854	

Fig.7.2 Ecopath - Basic estimate, Romania

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

	Group name	Trophic level	Habitat area (fraction)	Biomass in habitat area (t/km²)	Biomass (t/km²)	Z (/year)	Production / biomass (/year)	Consumption / biomass (/year)	Ecotrophic efficiency	Production / consumption
1	cetaceans	4.06386	1.00000	0.26440	0.26440		0.20000	4.10000	0.00000	0.04878
⊟	turbot									
2	turbot1	3.00000	1.00000	0.010450	0.010450	0.65000		4.06602	0.15960	0.15986
3	turbot2	3.08000	1.00000	0.067878	0.067878	0.65000		2.10923	0.30544	0.30817
4	turbot3	3.19200	1.00000	0.10819	0.10819	0.64000		1.48231	0.60972	0.43176
5	turbot4	3.88671	1.00000	0.10464	0.10464	0.64000		1.21555	0.97125	0.52651
6	turbot5	3.94920	1.00000	0.079982	0.079982	0.63500		1.07604	0.94778	0.59013
7	turbot6	4.07591	1.00000	0.053877	0.053877	0.63000		0.99444	0.98304	0.63352
8	turbot7	3.98503	1.00000	0.036922	0.036922	0.41000		0.94274	0.11826	0.43490
9	turbot8	4.03437	1.00000	0.027390	0.027390	0.40000		0.90975	0.00000	0.43968
10	turbot9	4.07302	1.00000	0.065808	0.065808	0.39000		0.86000	0.00000	0.45349
⊟	anchovy									
11	anchovy1	2.25000	1.00000	2.29316	2.29316	1.80000		10.5461	0.00381	0.17068
12	anchovy2	2.58000	1.00000	4.78000	4.78000	2.30000		5.54291	0.01525	0.41494
13	anchovy3	2.68000	1.00000	1.93932	1.93932	1.70000		3.76142	0.03797	0.45196
14	anchovy4	2.90000	1.00000	0.65661	0.65661	1.65000		3.08945	0.06577	0.53408
15	anchovy5	2.97000	1.00000	0.24202	0.24202	1.60000		2.69400	0.13636	0.59391
⊟	sprat									
16	sprat1	2.47000	1.00000	1.72544	1.72544	2.90000		13.4880	0.01662	0.21501
17	sprat2	2.80000	1.00000	3.50000	3.50000	1.63000		6.29489	0.08652	0.25894
18	sprat3	2.86000	1.00000	2.23973	2.23973	1.62000		4.32892	0.16004	0.37423
19	sprat4	2.90000	1.00000	0.83952	0.83952	1.61000		3.51580	0.25763	0.45793
20	sprat5	2.96000	1.00000	0.36843	0.36843	1.45000		3.00800	0.27293	0.48205
⊟	whiting									
21	whiting1	2.75000	1.00000	0.10273	0.10273	1.60000		9.86057	0.29304	0.16226
22	whiting2	2.89800	1.00000	0.30800	0.30800	1.52900		5.12981	0.11251	0.29806
23	whiting3	3.11000	1.00000	0.21225	0.21225	1.38200		3.61024	0.17089	0.38280
24	whiting4	3.21590	1.00000	0.089875	0.089875	1.48700		3.00662	0.35654	0.49457
25	whiting5	3.28836	1.00000	0.029603	0.029603	1.34700		2.69749	0.76788	0.49935
26	whiting6	3.35113	1.00000	0.013212	0.013212	1.34700		2.49000	0.63847	0.54096
⊟	gobies									
27	goby1	3.14800	1.00000	0.023886	0.023886	1.10000		6.37179	0.86292	0.17264
28	goby2	3.21840	1.00000	0.12641	0.12641	0.85000		3.22731	0.72911	0.26338
29	goby3	3.37160	1.00000	0.17600	0.17600	0.78000		2.23873	0.67239	0.34841
30	goby4	3.41750	1.00000	0.44169	0.44169	0.70000		1.60000	0.20520	0.43750
31	Mussels	2.10000	1.00000	22.0000	22.0000		2.97000	8.00000	0.01517	0.37125
32	Zoobenthos	2.32000	1.00000	1.02100	1.02100		2.50000	3.50000	0.78974	0.71429
33	Zooplankton	2.00000	1.00000	37.2900	37.2900		30.0000	60.0000	0.08055	0.50000

Fig.7.3 Ecopath - Basic estimate, West

	Group name	Trophic level	Habitat area (fraction)	Biomass in habitat area (t/km²)	Biomass (t/km²)	Z (/year)	Production / biomass (/year)	Consumption / biomass (/year)	Ecotrophic efficiency	Production / consumption
1	cetaceans	4.06386	1.00000	0.060000	0.060000		0.20000	4.10000	0.00000	0.04878
⊟	turbot									
2	turbot1	3.00000	1.00000	0.97260	0.97260	2.14070		13.5441	0.00012	0.15805
3	turbot2	3.08000	1.00000	2.11228	2.11228	2.03211		6.80578	0.00053	0.29859
4	turbot3	3.19200	1.00000	0.97409	0.97409	1.88624		4.62787	0.00386	0.40758
5	turbot4	3.88671	1.00000	0.30297	0.30297	1.66437		3.74279	0.02002	0.44469
6	turbot5	3.94920	1.00000	0.10566	0.10566	1.05171		3.28252	0.08020	0.32040
7	turbot6	4.07591	1.00000	0.047396	0.047396	1.02890		3.02993	0.14298	0.33958
8	turbot7	3.98503	1.00000	0.019988	0.019988	1.01563		2.87257	0.01158	0.35356
9	turbot8	4.03437	1.00000	0.0080878	0.0080878	1.01086		2.77025	0.00000	0.36490
10	turbot9	4.07302	1.00000	0.0051231	0.0051231	1.01041		2.67393	0.00000	0.37787
⊟	anchovy									
11	anchovy1	2.25000	1.00000	5.93694	5.93694	1.39400		10.6171	0.00024	0.13130
12	anchovy2	2.58000	1.00000	22.0194	22.0194	1.29160		5.47047	0.00061	0.23610
13	anchovy3	2.68000	1.00000	19.2437	19.2437	1.21600		3.82286	0.00118	0.31809
14	anchovy4	2.90000	1.00000	10.6327	10.6327	1.14850		3.14297	0.00308	0.36542
15	anchovy5	2.97000	1.00000	8.04222	8.04222	1.11354		2.69400	0.00383	0.41334
⊟	sprat									
16	sprat1	2.47000	1.00000	4.23253	4.23253	1.90000		13.1054	0.01563	0.14498
17	sprat2	2.80000	1.00000	12.1369	12.1369	1.62304		6.53156	0.00690	0.24849
18	sprat3	2.86000	1.00000	8.43842	8.43842	1.47694		4.44812	0.01108	0.33204
19	sprat4	2.90000	1.00000	4.02072	4.02072	1.25904		3.58410	0.02613	0.35129
20	sprat5	2.96000	1.00000	2.93107	2.93107	1.14599		3.00800	0.03788	0.38098
⊟	whiting									
21	whiting1	2.75000	1.00000	0.10646	0.10646	1.00000		9.51296	0.86710	0.10512
22	whiting2	2.89800	1.00000	0.37947	0.37947	1.52900		5.12981	0.17056	0.29806
23	whiting3	3.11000	1.00000	0.26150	0.26150	1.38200		3.61024	0.26423	0.38280
24	whiting4	3.21590	1.00000	0.11073	0.11073	1.48700		3.00662	0.55618	0.49457
25	whiting5	3.28836	1.00000	0.036472	0.036472	1.34700		2.69750	0.81649	0.49935
26	whiting6	3.35113	1.00000	0.016278	0.016278	1.34700		2.49000	0.33777	0.54096
⊟	gobies									
27	goby1	3.14800	1.00000	0.076971	0.076971	1.45850		6.19120	0.80973	0.23558
28	goby2	3.21840	1.00000	0.29979	0.29979	1.25480		3.12351	0.45736	0.40173
29	goby3	3.37160	1.00000	0.29453	0.29453	1.11248		2.14474	0.52395	0.51870
30	goby4	3.41750	1.00000	0.34084	0.34084	1.09845		1.61000	0.46979	0.68227
31	Mussels	2.10000	1.00000	5.50000	5.50000		4.97541	10.00000	0.86697	0.49754
32	Zoobenthos	2.32000	1.00000	1.65545	1.65545		38.0000	70.0000	0.14465	0.54286
33	Zooplankton	2.00000	1.00000	9.30000	9.30000		45.0000	60.0000	0.89812	0.75000

Fig.7.4 Ecopath - Basic estimate, All

Ecosim use of mass-balance results (from Ecopath) for parameter estimation. For a time dynamic simulation with Ecosim, we use time series data from 2007 to 2013; and duration of simulation is 14 years(2007 - 2020). Data used for 'Time series grid': biomass, catches, IUU(Illegal,Unreported and Unregulated) - only for turbot, fishing effort(GNS, OTM, FPN, FPO, LLS, LHP, SB,TBB). The time series fitting use fishing effort data as driving factors for the Ecosim model runs.

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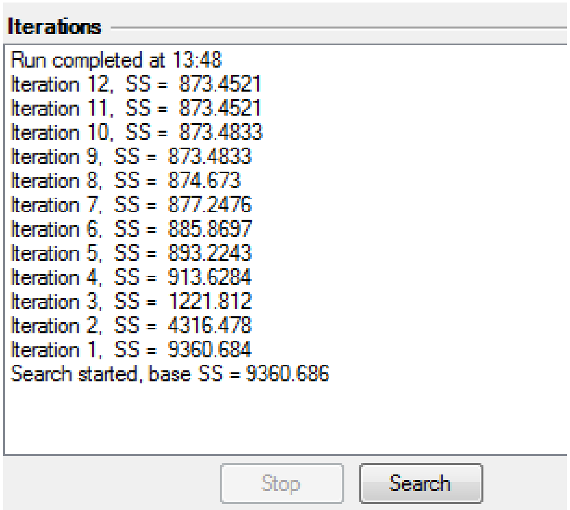

Fig.7.5 Ecosim – Iterations, Romania

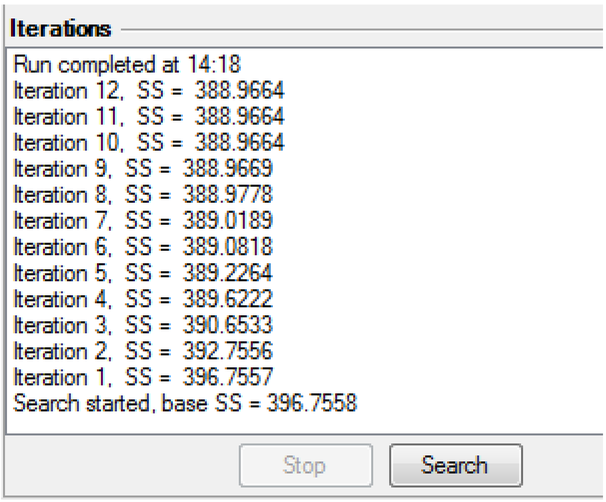

Fig.7.6 Ecosim – Iterations, West

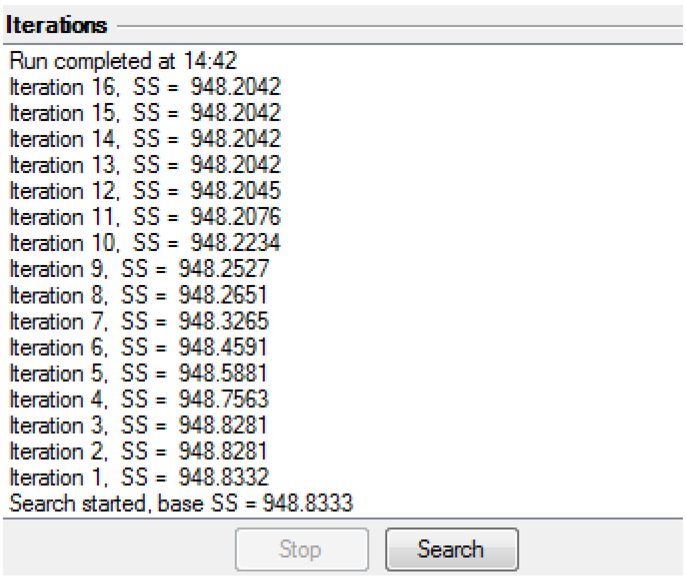

Fig.7.7 Ecosim – Iterations, All

One key feature of Ecosim is its ability to allow exploring the implications on system dynamics of different views of how the biomass of different groups in ecosystem is controlled. The two extreme views are 'predator control' (also called top-down control) and 'prey (or bottom-up) control'. We model this using 'vulnerabilities', which represent the factor that a large increase in predator biomass will cause in predation mortality for a given prey. Low vulnerability (close to 1) means that an increase in predator biomass will not cause any noticeable increase in the predation mortality the predator may cause on the given prey. A high vulnerability, e.g., of 100, indicates that if the predator biomass is for instance doubled, it will cause close to a doubling in the predation mortality it causes for a given prey.

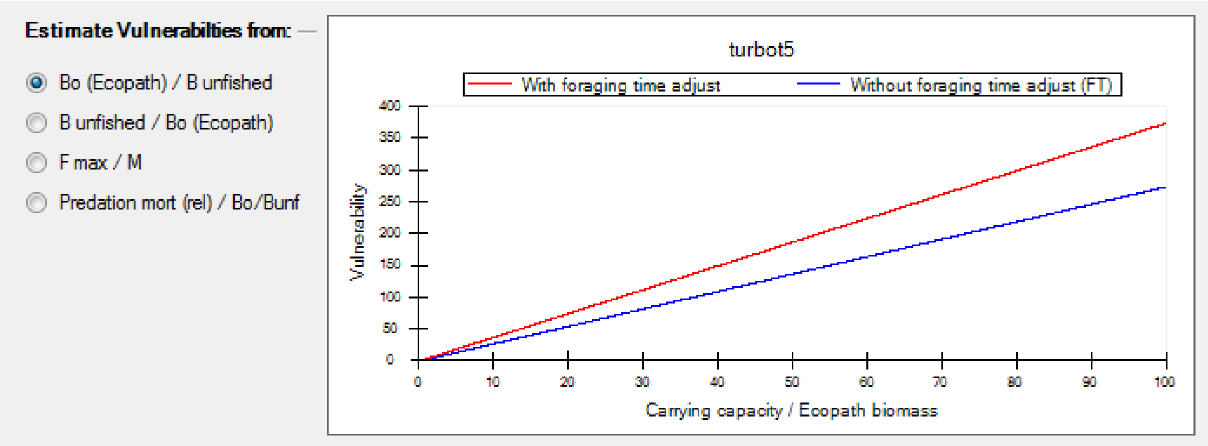

Fig.7.8 Ecosim – Vulnerabilities(1), Romania

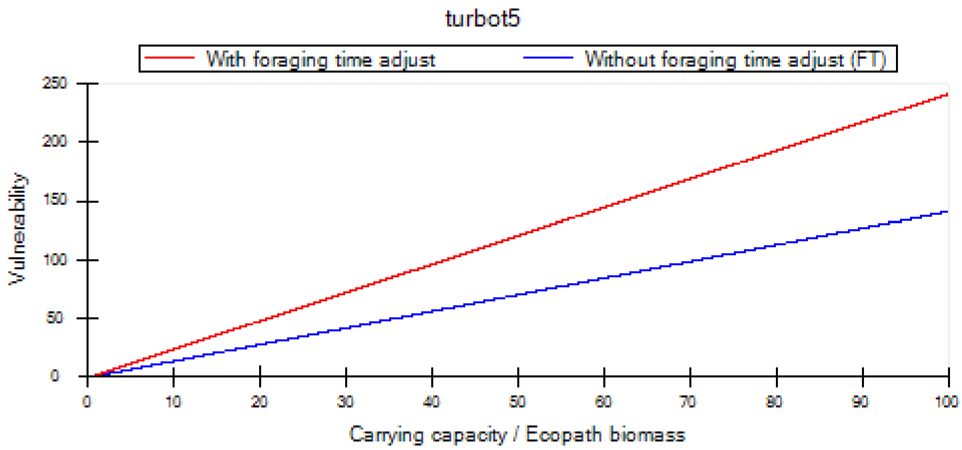

Fig.7.9 Ecosim - Vulnerabilities(1), West

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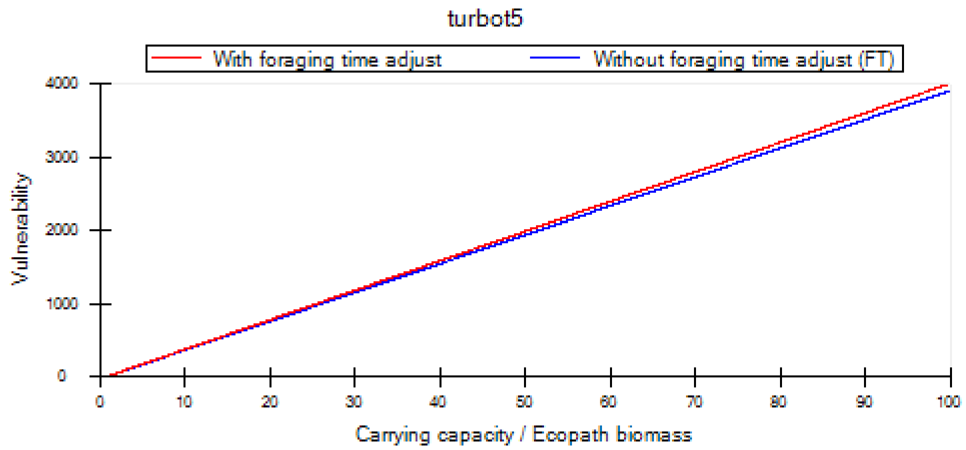

Fig.7.10 Ecosim - Vulnerabilities(1), All

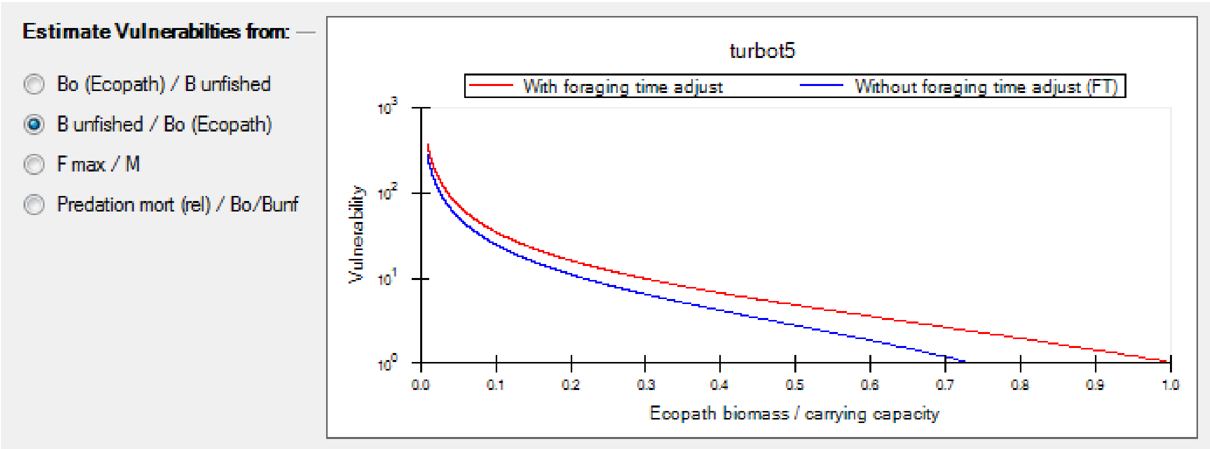

Fig.7.11 Ecosim - Vulnerabilities(2), Romania

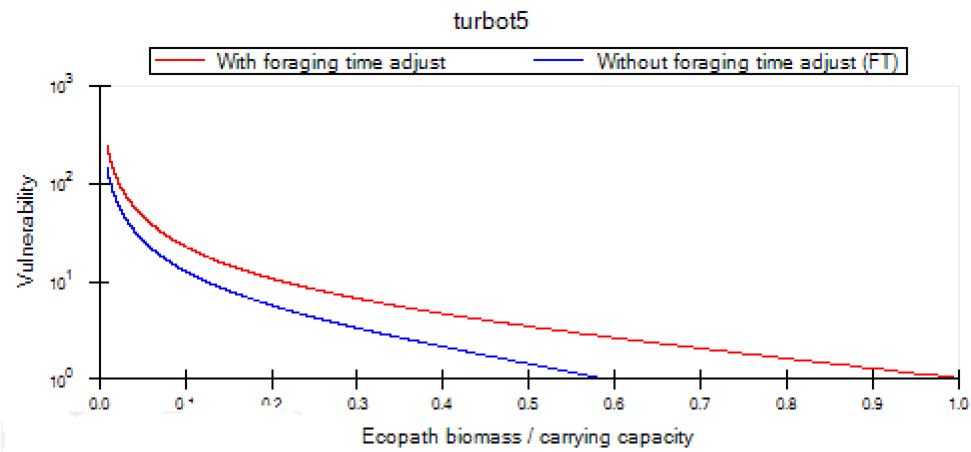

Fig.7.12 Ecosim - Vulnerabilities(2), West

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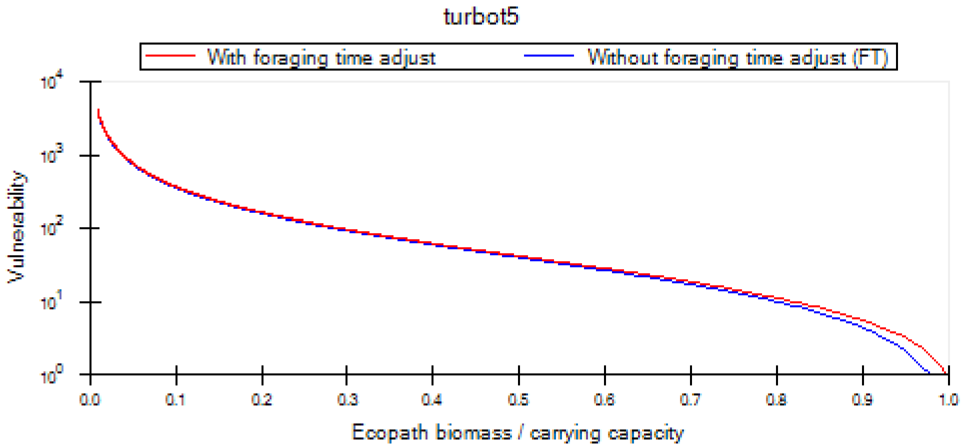

Fig.7.13 Ecosim - Vulnerabilities(2), All

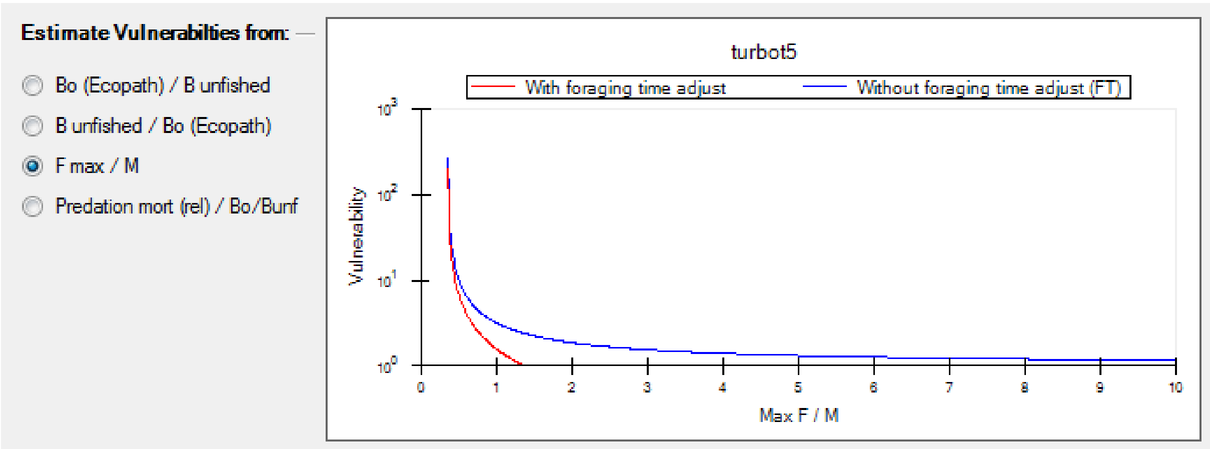

Fig.7.14 Ecosim - Vulnerabilities(3), Romania

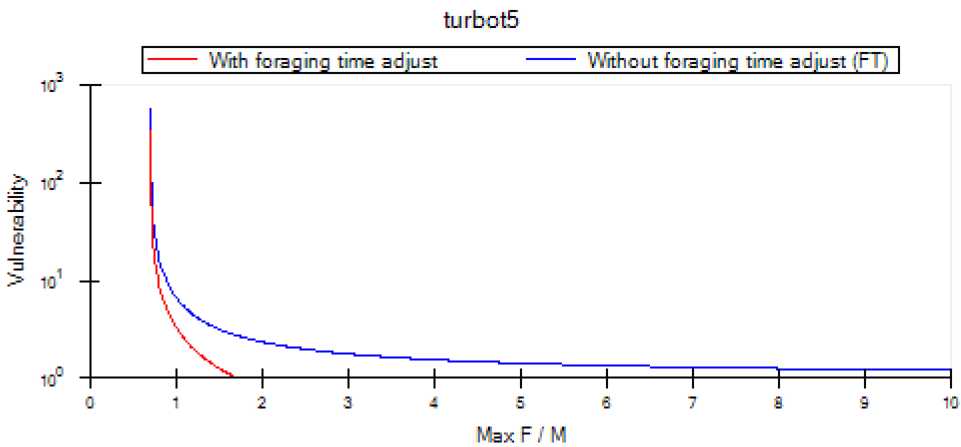

Fig.7.15 Ecosim - Vulnerabilities(3), West

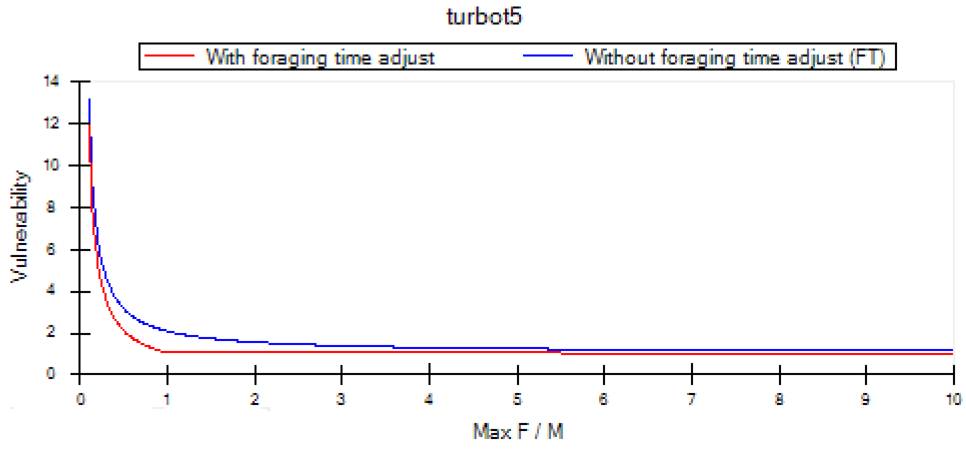

Fig.7.16 Ecosim - Vulnerabilities(3), All

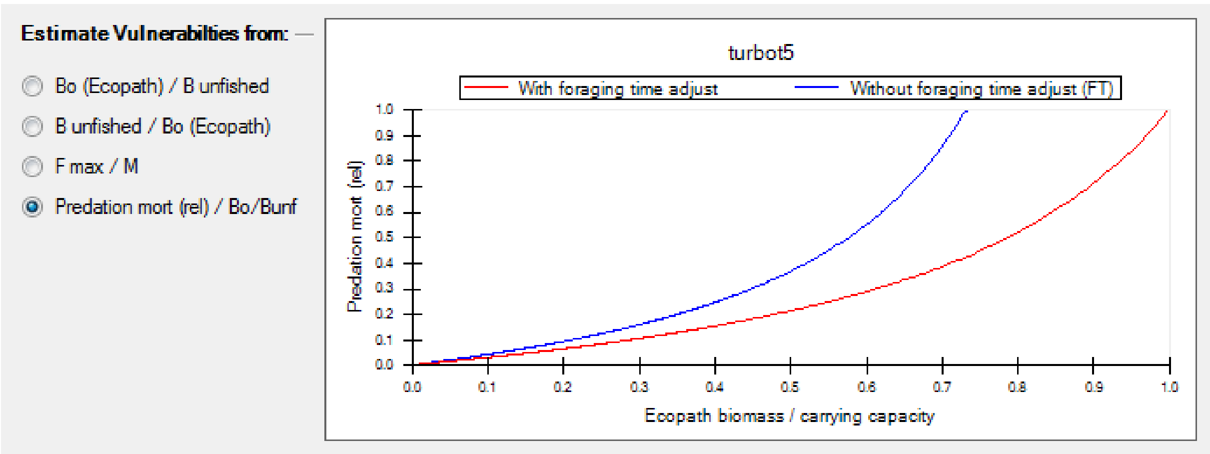

Fig.7.17 Ecosim - Vulnerabilities(4), Romania

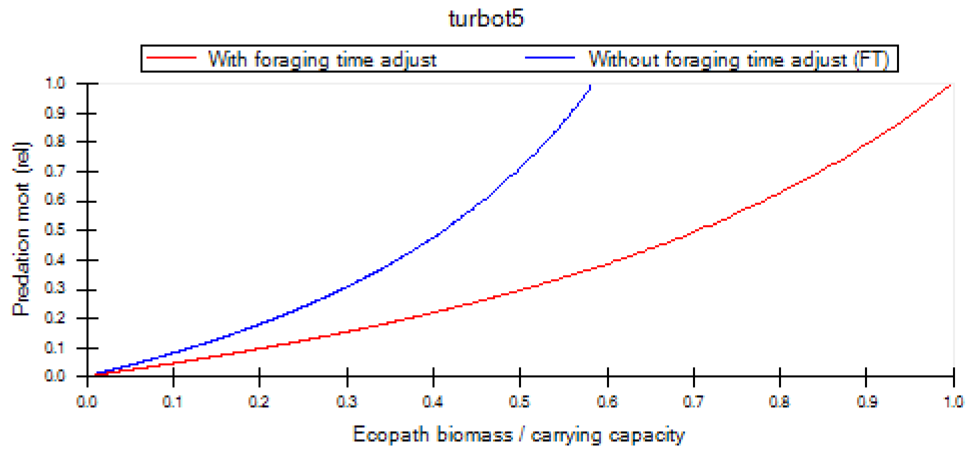

Fig.7.18 Ecosim - Vulnerabilities(4), West

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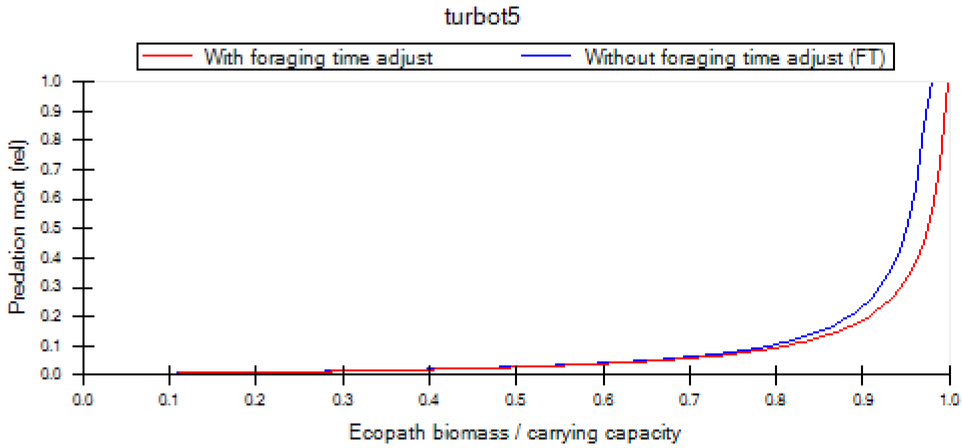

Fig.7.19 Ecosim - Vulnerabilities(4), All

The Run Ecosim form showing predicted (coloured lines) and observed (coloured dots) biomass trajectories. The red fishing mortality sketch pad can be seen in the bottom panel. The 'Fisheries' fleet has been selected from the drop-down Target menu.

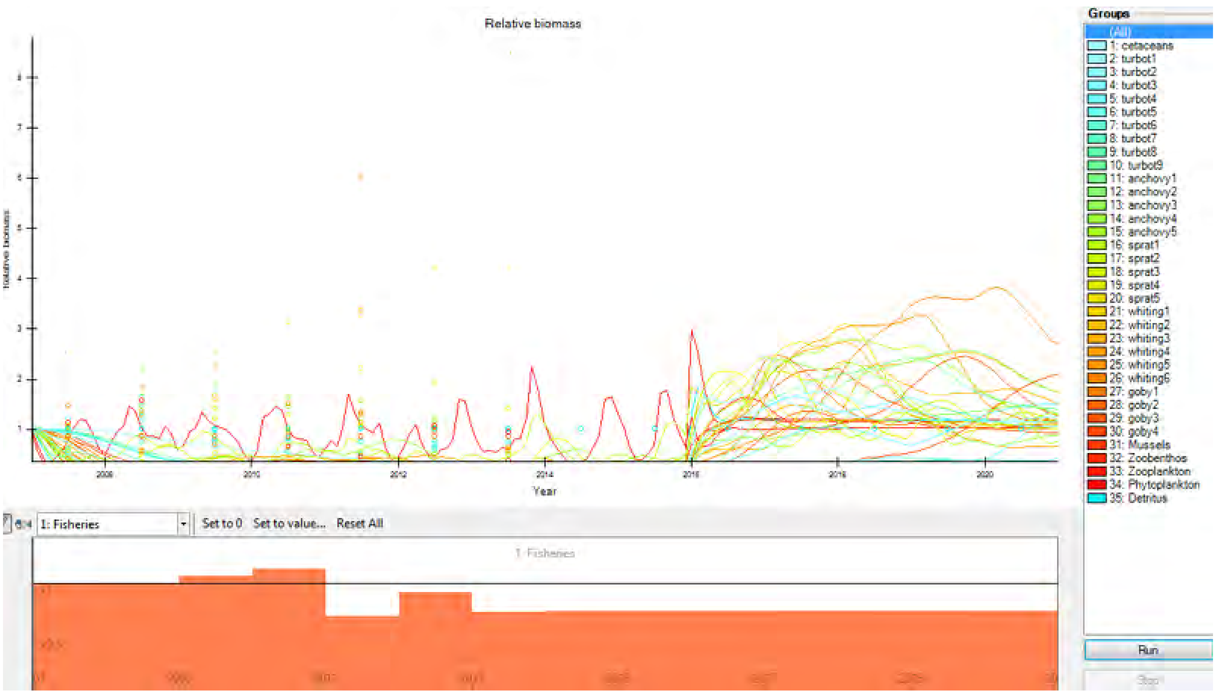

Fig.7.20 Run Ecosim, Romania

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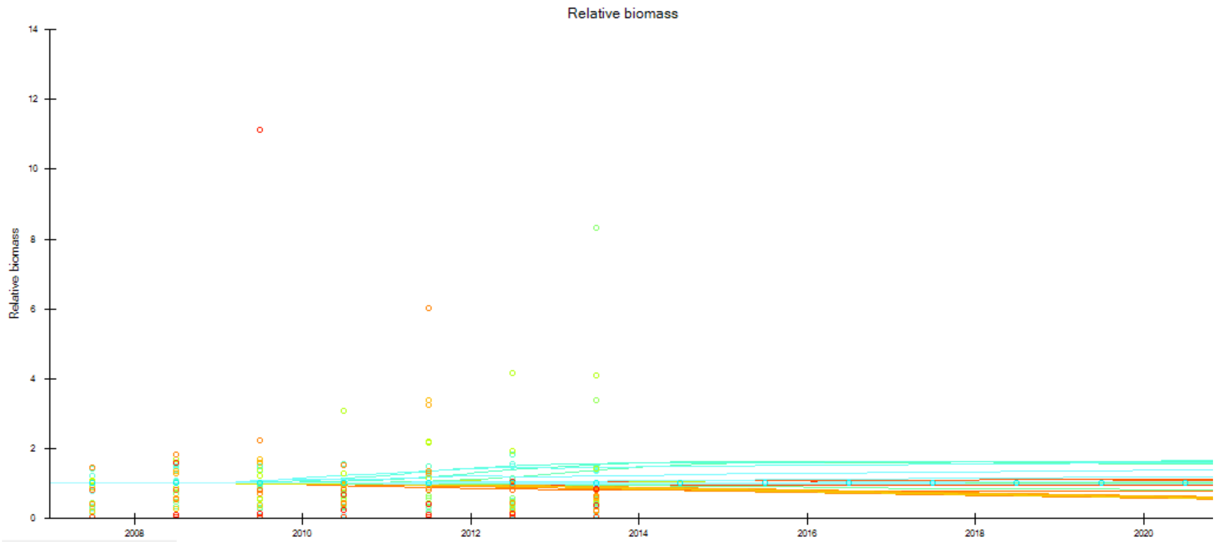

Fig.7.21 Run Ecosim, West

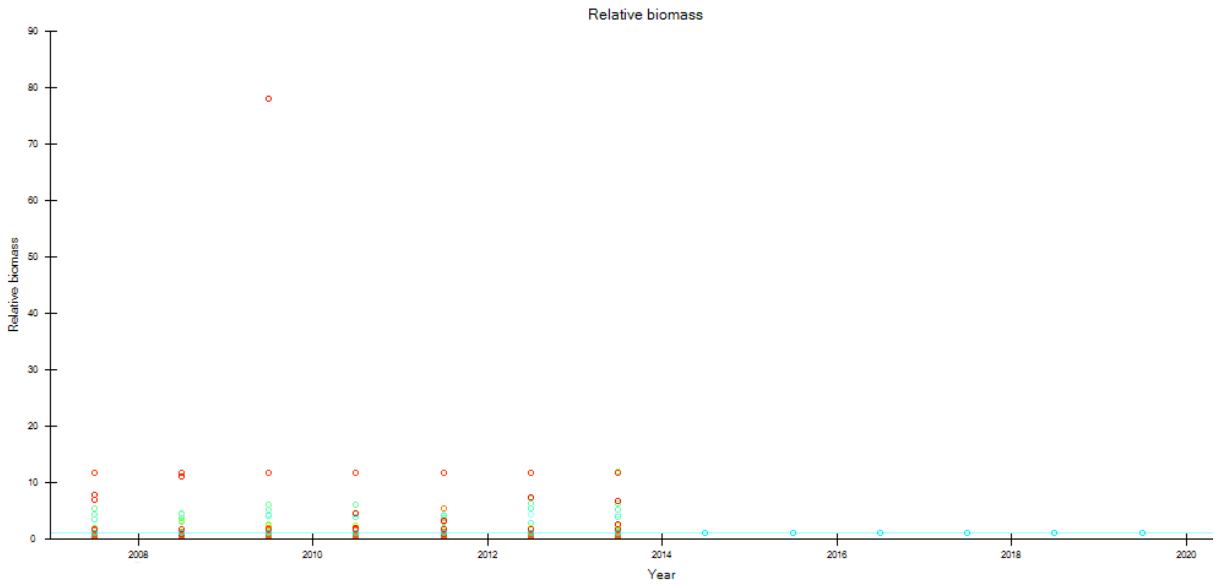

Fig.7.22 Run Ecosim , All

Selecting Ecosim plot after running Ecosim (see Run Ecosim) opens a form displaying a series of plots of the results of the Ecosim simulations. Select the group to be displayed by clicking on its name in the Groups window on the right of the form. Plots will be displayed showing time series of predicted biomass, consumption/biomass, predation mortality, total mortality, feeding time, percentage of prey, catch, average weight and fishing mortality (Fig. 23- 25).

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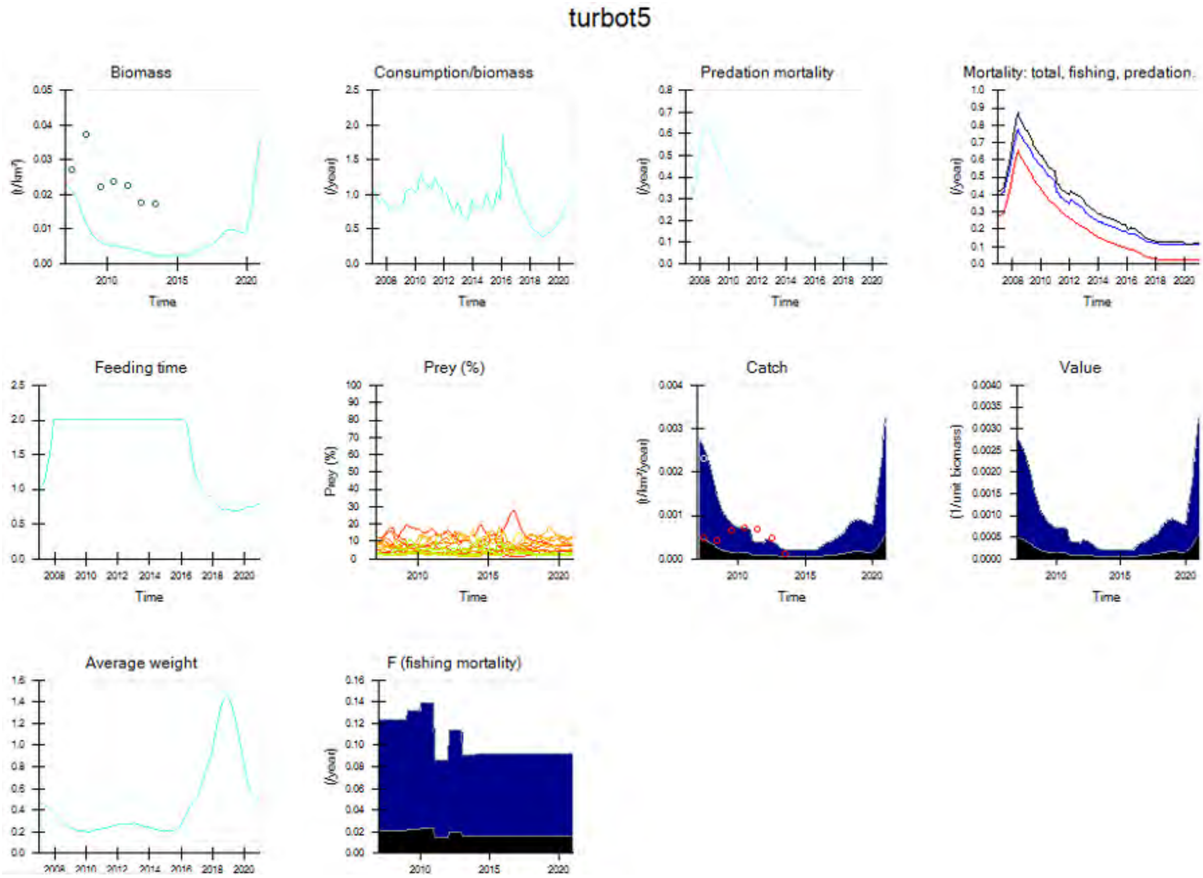

Fig.7.23 Ecosim group plots for Turbot age 5, Romania

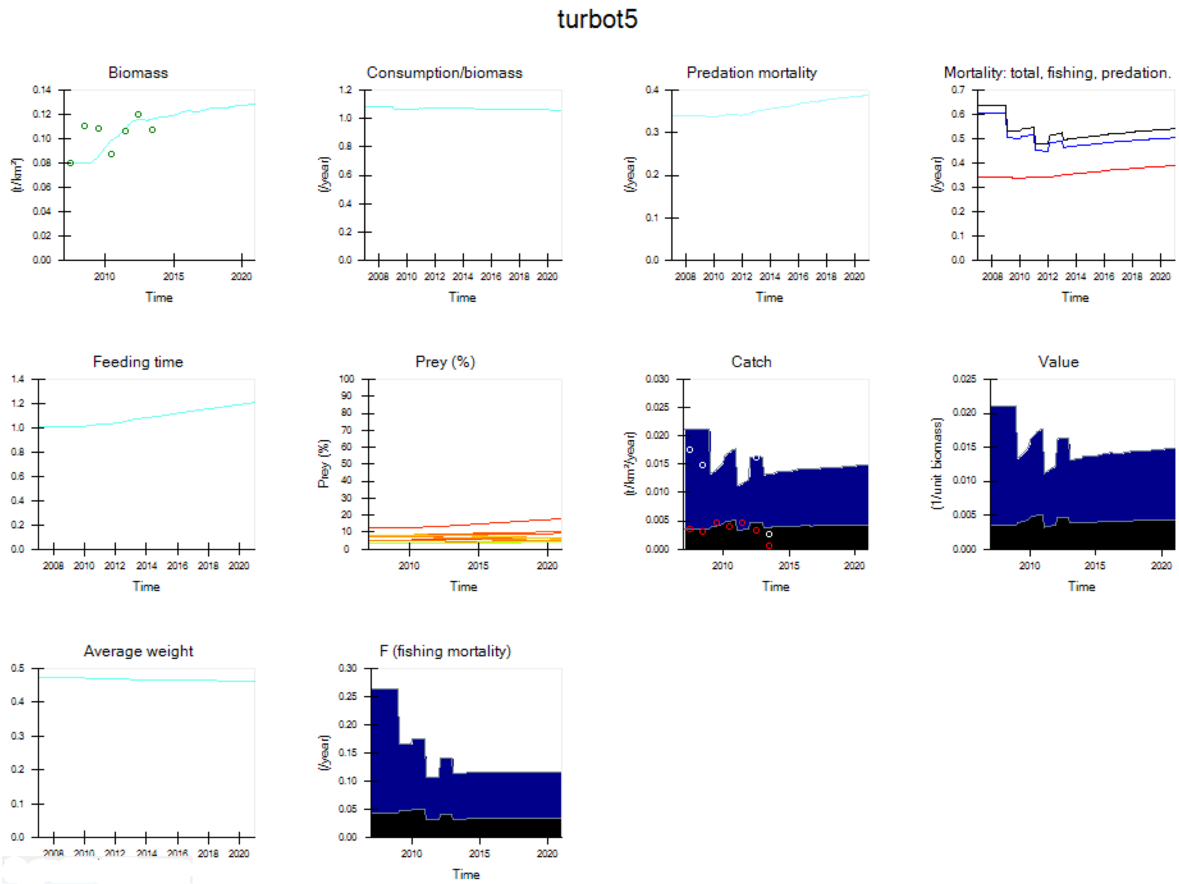

Fig.7.24 Ecosim group plots for Turbot age 5, West

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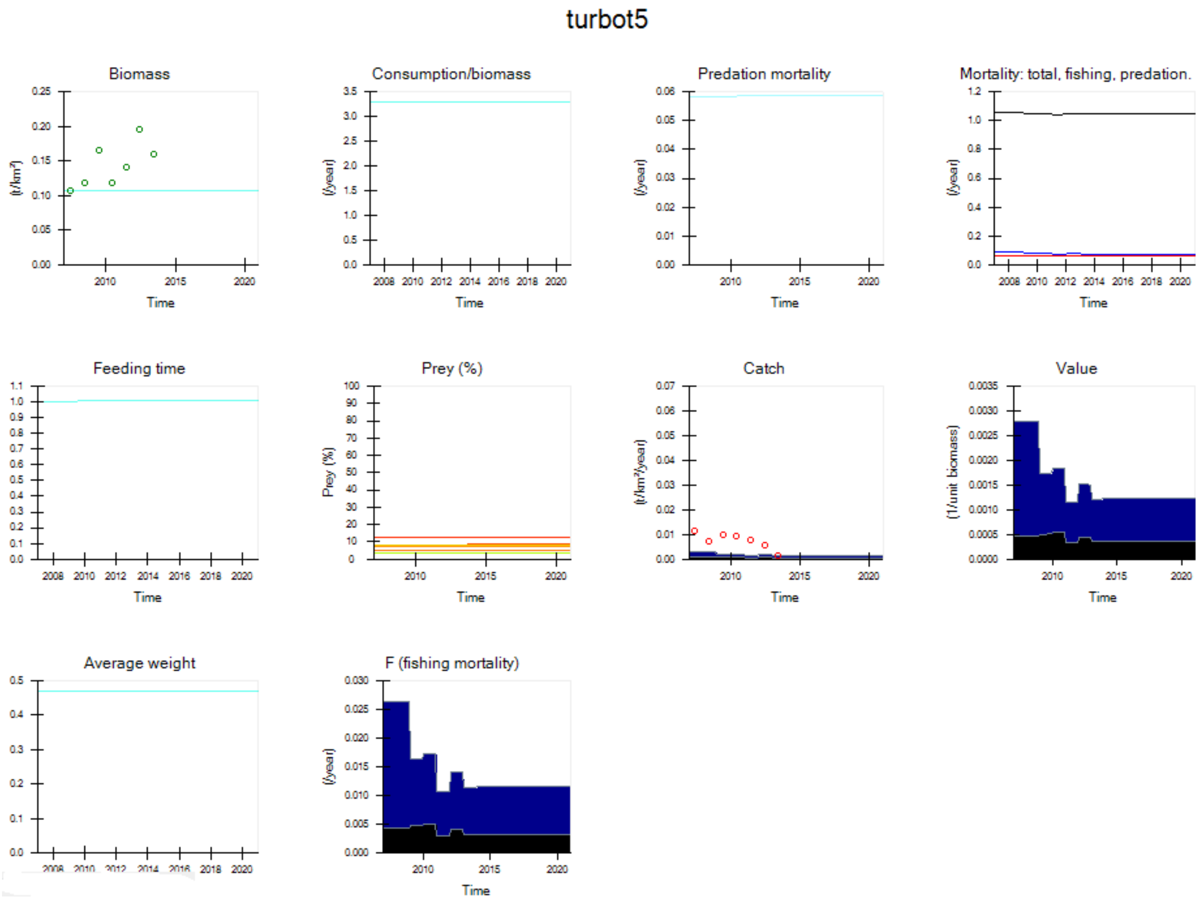

Fig.7.25 Ecosim group plots for Turbot age 5, All

As we can see in Figure 23-25, dots will appear on the Biomass plot showing the observed biomass time series, while dots on the Catch plot will show observed catches . Ecosim's predicted biomasses and catches are shown as lines.

Selecting 'Ecosim results' after running Ecosim shows a summary of results for the run, with start and end dates set using the vertical combo box in the 'Summary periods' of the 'Ecosim results' form.

Summary periods

Start year of first summary period:	0.00000
Start year of second summary period:	13.00000
Time steps in both summary periods:	12

Content

- ◉ Fleet
- ○ Group, landed by:
- ○ Indices

	Fleet name	Catch (start)	Catch (end)	Catch (E/S)	Value (start)	Value (end)	Value (E/S)	Cost (start)	Cost (end)	Cost (E/S)	Effort (E/S)
1	Fisheries	0.02187	0.04387	2.00564	0.02187	0.04387	2.00564	0.03596	0.02694	0.74911	0.74911
2	IUU	0.00984	0.01069	1.08666	0.00984	0.01069	1.08666	0.00979	0.00734	0.74911	0.74911
	Total	0.031710	0.054558	1.72053	0.031710	0.054558	1.72053	0.045754	0.034275	0.74911	

Fig.7.26 Ecosim results, Romania

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

Summary periods

Start year of first summary period: 0.00000

Start year of second summary period: 13.00000

Time steps in both summary periods: 12

Content

(•) Fleet
( ) Group, landed by:
( ) Indices

	Fleet name	Catch (start)	Catch (end)	Catch (E/S)	Value (start)	Value (end)	Value (E/S)	Cost (start)	Cost (end)	Cost (E/S)	Effort (E/S)
1	Fisheries	1.53657	1.17014	0.76152	1.53657	1.17014	0.76152	1.22927	0.92086	0.74911	0.74911
2	IUU	0.09324	0.04791	0.51388	0.09324	0.04791	0.51388	0.07459	0.02794	0.37455	0.37455
	Total	1.62981	1.21805	0.74736	1.62981	1.21805	0.74736	1.30386	0.94880	0.72768	

Fig.7.27 Ecosim results, West

Summary periods

Start year of first summary period: 0.00000

Start year of second summary period: 13.00000

Time steps in both summary periods: 12

Content

(•) Fleet
( ) Group, landed by:
( ) Indices

	Fleet name	Catch (start)	Catch (end)	Catch (E/S)	Value (start)	Value (end)	Value (E/S)	Cost (start)	Cost (end)	Cost (E/S)	Effort (E/S)
1	Fisheries	0.04495	0.03368	0.74919	0.04495	0.03368	0.74919	0.03956	0.02963	0.74911	0.74911
2	IUU	0.01224	0.00462	0.37775	0.01224	0.00462	0.37775	0.00979	0.00367	0.37455	0.37455
	Total	0.057193	0.038302	0.66969	0.057193	0.038302	0.66969	0.049350	0.033301	0.67478	

Fig.7.28 Ecosim results, All

References

Ahlbeck I, Hansson S, Hjerne O. 2012. Evaluating fish diet analysis methods by individual-based modelling. Can. J. Fish. Aquat. Sci. 69(7): 1184-1201

Bănaru, D., M. Harmelin-Viviena. 2009. Feeding behaviour of Black Sea bottom fishes: Did it change over time? , Acta Oecologica: 35, 6: 769-777

Villy Christensen, Carl J. Walters, Daniel Pauly and Robyn Forrest. 2008. Ecopath with Ecosim version 6, User Guide. http://www.boblme.org/documentRepository/Ecopath%20parameters.pdf

8. Chatham Rise case study

Brief description of the case study objectives

The Chatham Rise is a broad submarine ridge about 800 km long and 300 km wide that extends eastwards from New Zealand landmass into the southwest Pacific Ocean (Fig 8.1). High phytoplankton abundance in this region is a conspicuous feature of ocean colour images of the Southern Ocean (e.g. Gordon et al. 1986; Banse & English 1997; Murphy et al. 2001). Elevated phytoplankton productivity is attributed to the presence of the Subtropical Front (STF) being bathymetrically locked to the Chatham Rise (Murphy et al. 2001; Sutton, 2001; Uddstrom & Oien, 1999). The STF above the Chatham Rise forms part of a 25,000 km-long convergence zone of northern Subtropical (ST) waters, and southern Subantarctic (SA) waters that encircles the globe. The mixing of nitrate-depleted ST water, with nitrate-rich SA water in the Chatham Rise region leads to elevated phytoplankton productivity (Boyd et al. 1999). Elevated oceanic productivity here is responsible for supporting the complex and valuable Chatham Rise ecosystem, including deep-water fisheries (e.g. orange roughy, oreo, hoki), an unusually rich benthic ecosystem, as well as seabird and marine mammal populations. The STF over the Chatham Rise is an area of vigorous mixing and eddy activity (e.g. Heath 1976; Belkin 1988; Uddstrom & Oien 1999; Stanton 1997; Chiswell 1994; Sutton 2001).

For the purposes of the modelling work, we define the study area as occupying the region bounded by the 250 m depth contour to the west (edge of the continental shelf) and the 1250 m depth contour elsewhere. The contours are linked at 172°E (SW corner) and 43°S (NW corner). This region has an area of approximately 222,800 km2. The mean depth of the region is 620 m. The Chatham Island group (close to 176.5°W) have an area of only 960 km^2 (<0.5% of the study region).

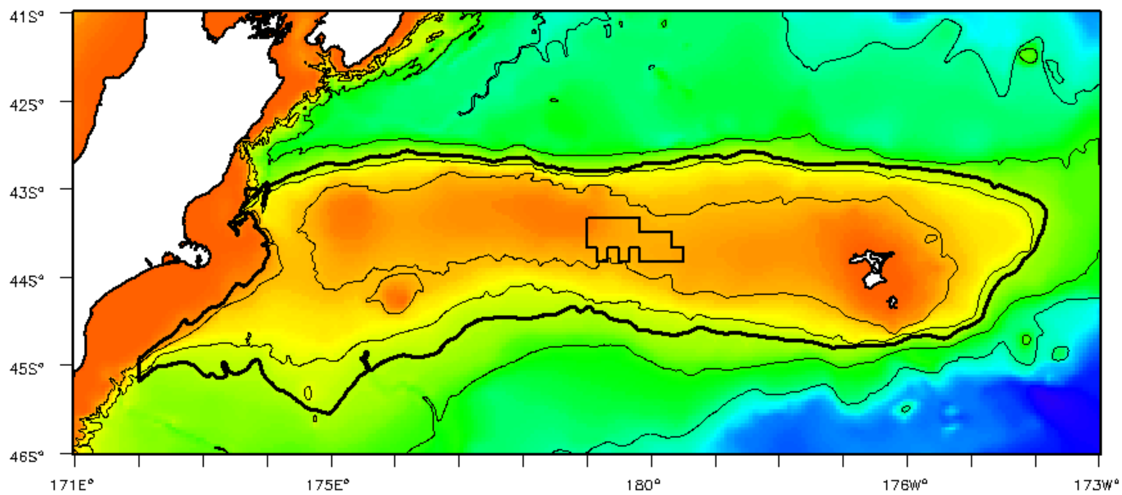

Figure 8.8. Depth of water over the Chatham Rise. High values are red; low values are blue (range 0–5100 m). The trophic model area and prospecting licence area are shown as thick black outlines. Depth contours (thin

black lines) are plotted at 500, 1000, 2000 and 3000 m. The polygon within the Chatham Rise represents the proposed seabed mining area.

The ecosystem supports substantial commercial fisheries production (60% of New Zealand‘s total landings), and also a high diversity of seabirds, cetacean, and large pelagic fish species, many of which are protected under New Zealand law but threatened by human activities. There are also protected deepwater corals, and a number of Benthic Protection Areas and Seamount Closures in the region (Helson et al. 2009), designed to protect seaded biodiversity.

Recently a proposal to extract phosphorite nodules from the seabed along part of the crest of the Chatham Rise was reviewed by New Zealand‘s Environmental Protection Agency, and the work described here formed part of the evaluation of the potential effects of seabed mining, to inform the Agency‘s decision (http://www.epa.govt.nz/EEZ/previous-activities/notified-consents/chatham_rock_phosphate/Pages/default.aspx).

Modelling framework

A balanced model of the food web of the Chatham Rise was developed by bringing together information on all biota in the ecosystem. The model is static (rather than dynamic like EwE or Atlantis), and provides information at the spatial scale of the whole Chatham Rise and averaged over a typical year and provides information about trophic connections only; provision of habitat is not considered by the model. The model quantifies the transfer of organic material through a food web based on the widely used mass-balance identities of the Ecopath trophic model (Christensen & Walters 2004; Christensen & Pauly 2002). Key information includes main species, their biomass, energetics (feeding and growth rates; assimilation efficiencies) and diets. The model has 37 groups: seabirds, toothed whales & dolphins, baleen whales, seals; 9 demersal fish groups; 4 mesopelagic groups; 10 groups of benthic invertebrates; 3 groups of small zooplankton; phytoplankton, bacteria (2 groups) and detritus (3 groups).

The trophic model uses the semi-objective balancing method first described in Pinkerton et al. (2010) and subsequently refined (Pinkerton, 2011b). Each of the model parameters initially estimated has an associated uncertainty because the values are imperfectly and incompletely observed, and because the parameters vary between years and hence differ from our modeled average recent year. Preliminary estimates of all parameters are adjusted to obtain a model where all the equality constraints are fulfilled, minimising a „cost function" which provides a measure of the overall amount of change to parameters needed to achieve balance, taking into account relative uncertainties between parameters and ensuring even adjustment across groups in the model.

Two kinds of sensitivity tests to undertainty were conducted, sensitivity to the balancing process, and sensitivity to the initial parameter estimates. To test the sensitivity of the model results to the balancing method, the relative uncertainty factors for biomass, productivity rate and diets of each group in the model were multiplied by a random factor representing changes of up to a factor of 5, with an increase or decrease equally likely. The model was then balanced using the new relative uncertainty factors and mixed trophic impact analysis was carried out to ascertain the new index of trophic importance for each group. This exercise was repeated 2000 times. To test the sensitivity of the baseline model results to initial parameter estimates, starting with biomass, the initial estimates were multiplied by a random factor representing changes of up to a factor of 5, with an increase or decrease equally likely. The model was then balanced using the new relative uncertainty factors and mixed trophic impact (analysis was carried out to ascertain the new index of trophic importance for each group. This exercise was repeated 2000 times. An extended description of this model can be found in the Deliverable 4.1.

The structure of the of the foodweb of the Chatham Rise as represented in the balanced trophic model is presented in Fig. 8.2. Biomasses of groups in the model varied by 4.3 orders of magnitude, between 2.9 gC m^{-2} (phytoplankton) and 0.13 mgC m^{-2} (holothurians). Total net primary production (NPP) in the balanced model was 276 gC m^{-2} y^{-1}. This was slightly below the mean for 14 large marine ecosystems around the world which support significant large-scale fisheries (encompassing temperate boreal shelves and eastern boundary currents; Conti & Scardi (2010)).

On the basis of this model, the average trophic importance (sensu Libralato et al. 2006) of the species or model groups has been calculated form the Mixed Trophic Impact matrix. Trophic importance (TI) is a measure of the overall effect on food-web structure of changes to the abundance of one group in the model. “Trophic importance” is preferred over “ecological importance” as only trophic effects are considered by the analysis. This measure is preferred over “keystoneness” since the meaning of the latter has become confused. Keystoneness was defined by Power et al. (1996) as the amount by which the trophic importance of a species exceeds that expected on the basis of abundance alone. Other interpretations of keystoneness essentially equate it to trophic importance (Libralato et al. 2006). In any case, trophic importance is the relevant measure in terms of assessing by how much changes in the abundances of species caught by the fishery are likely to affect the food-web, irrespective of whether those species have high or low biomass in the ecosystem.

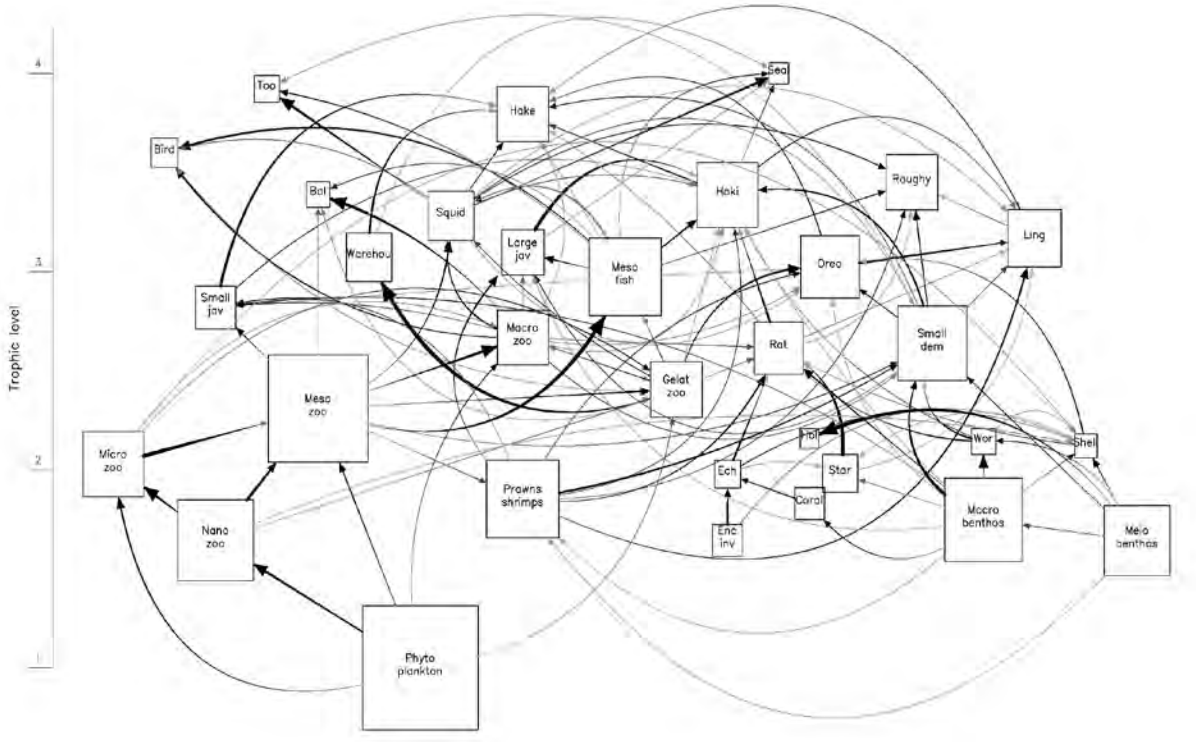

Fig. 8.2. Food-web model flow diagram, with arrows showing the direction of organic carbon flow (Pinkerton, 2013). Bacterial and detrital groups omitted for clarity. Bigger boxes mean more biomass. Boxes are positioned vertically according to trophic level. Thick/dark lines show higher flows in or out of the group. Too = toothed whales; Bal = baleen whales; Ech = echinoids; Star = Seastars and brittlestars; Shell = shelled megabenthos; Hol = holothurians (sea cucumber); Enc inv = encrusting benthic invertebrates; Wor = large benthic worms; Rat = rat-tail fish; Jav = javelin fish; Gelat zoo = gelatinous zooplankton (salps mainly); Meso fish = mesopelagic (midwater) fish; Small dem = small demersal fish.

Analysis

The potential implications of seabed mining on the Chatham Rise were examined though initial examination of the trophic importance, and then qualitative (expert option) assessment of the anticipated direct impacts of mining on the groups with the highest trophic importance. Within this modelling framework it was not possible to develop a quantitative estimate of the ecosystem effects of mining based on combining the anticipated direct, habitat-mediated and indirect (trophic) effects of mining. Some information required to do this is not known (e.g. to what extent some species are dependent on habitat in the proposed mining area for spawning/survival of early life stages; the types of control affecting the abundances of species).

Estimates of trophic importance (Fig. 8.3.) were generally not sensitive to the balancing method or initial parameters. The rank trophic importance was very consistent across the sensitivities, and the model is considered robust to to the balancing methodology and uncertainties in the initial estimation of biomasses.

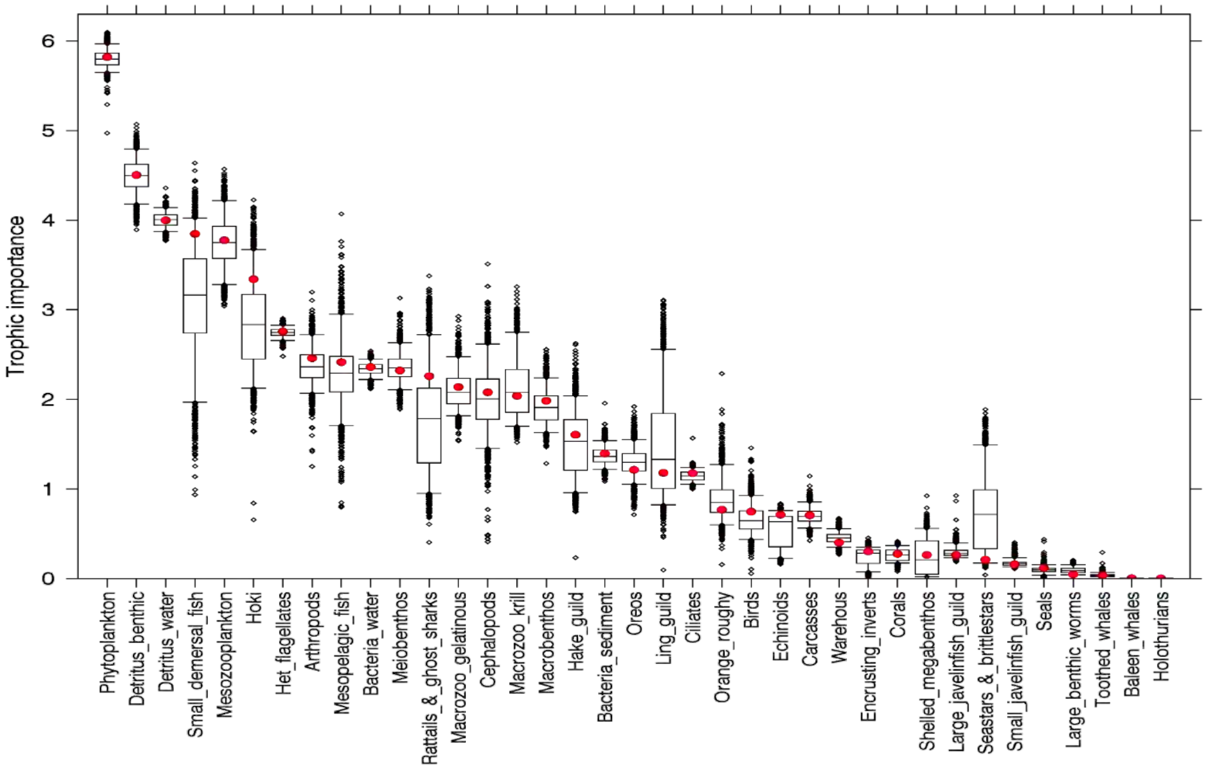

Fig. 8.3. Sensitivity of trophic importance of model groups to the balancing method. Groups are arranged according to decreasing trophic importance in the baseline model (Pinkerton, 2013 and red dots). Boxes show the effect of randomly varying the uncertainty parameters by up to a factor of 5 (changes between factor of 0.2 and 5 from baseline model), N=1517. Boxes show 25th–75th percentiles (with median line); whiskers show 5th–95th percentiles; individual outliers shown as black dots.

Mortalities caused by the proposed mining on benthic species have been estimated by assuming that all removed biota are killed by the processing. If it is assumed that all life stages of a given species have the same spatial distribution, the direct impact of the mining of that species depends on the proportion of biomass that is removed, or, approximately, the proportion of the spatial range of the organism that is mined.

Mining is expected to generate a sediment plume that remains relatively close to the seabed. This plume will come into contact with some organisms but its effect on them is hard to estimate because the edge of the plume is not easy to define as there will be a continuum of sediment concentrations that will vary in space and time, the exposure of various organisms to the plume is hard to predict as we do not know their horizontal or vertical distributions, and the effect of the plume on different organisms is not well known.

Notwithstanding these complexities, to a first approximation, if all life-stages of a given species have the same spatial distribution, the impact of the plume on species depends on how much of their biomass occur in the area where the plume is. The overall direct effect of mining on a species then depends on the effect on each life stage and on how important that life stage is to the overall productivity of the species. For example, if there was a “hotspot” of spawning and/or juveniles close to the mining area, the mining could have a large effect on the species even though the main adult range is elsewhere.

The direct impact of mining on some species depends on how much biomass of that species occurs in the area affected by mining. For a species that occurs largely in the area to be mined/affected by the sediment plume, the impact is likely to be substantial (e.g. corals). For species that occur throughout the Chatham Rise, with no particular biomass "hotspot" of any life stage close to the mining area (e.g. hoki) the direct effects of mining are likely to be low.

The spatial distribution of spawning/juveniles for many other groups, and especially for some species of demersal, mesopelagic and hyperbenthic organisms, are not known. Most mesopelagic organisms (e.g. myctophids) are likely to spawn in the midwater which makes it less likely that the mining itself or the plume will affect them. There is some suggestion in the acoustic data of a krill biomass hotspot near the mining area (Pinkerton, 2014), and spawning success of some krill is related to the benthos, but this is not well known for krill species on the Chatham Rise. Areas of spawning for some demersal fish have been investigated but many have not because of lack of data in some areas and some seasons (O'Driscoll et al., 2014) especially for species of demersal fish that are too small or too slender to be caught by research or commercial trawl gear.

Modelling of the spatial distribution of various benthic communities (Rowden et al., 2014) suggests a complex mosaic of different habitat-forming biota in the vicinity of the proposed mining. Suitable habitat may be ecologically-important for some organisms or for some life-stages of organisms. However, the roles of these benthic communities in providing habitat to demersal, mesopelagic, hyperbenthic and benthic species (or to some life-stages of these species) are not well known.

Changes to the abundance of one organism may affect its predators and prey, potentially leading to trophic cascades or even regime shift if the perturbation is great enough. The extent to which changes to one organism affect others through trophic interactions depends on the type of control (e.g. top-down, bottom-up, extrinsic), i.e. is the abundance of a species affected by how much it is predated, how much food it can find, or by a non-trophic factor such as the recruitment or settlement rate (how many adults enter the population each year). The relative importance of different factors in affecting the dynamics of species on the Chatham Rise is poorly known (as it is in marine ecosystems generally).

Mixed trophic impact analysis is a relatively simple method of investigating how important an organism might be to the overall food-web. The method essentially assumes that: (a) increasing the biomass of a predator will negatively impact its prey; (b) increasing the biomass of a prey will positively impact its predators; and (c) the magnitude of the impact is higher if the impacting group provides more food to, or consumes more of the production of, the impacted group. Trophic importances based on the Chatham Rise trophic model following sensitivity analysis are provided in Fig. 8.3. The estimated direct effects of mining on groups in the trophic model are given in Table 8.1. The table is arranged in order of decreasing trophic importance. For the purpose of considering potential impact of mining at the scale of the Chatham Rise food-web, more consideration should be given to groups at the top of the table than those lower down, and only the top-half (in terms of trophic importance) of groups are shown. The anticipated direct impacts of mining on 10 of the 11 groups with the highest trophic importances are likely to be low or negligible, because these groups are widely spread over the Chatham Rise or planktonic so the scale of impact is likely to be small. This analysis suggests that the four groups with trophic importances that are higher than average and are at the highest direct/habitat-mediated risk from mining are likely to be small demersal fish, hard-bodied macrozooplankton (krill), cephalopods and rattails & ghost sharks.

This conclusion is reached on the basis that some of the important species in these groups may depend on hard benthic habitat (such as that near to the proposed mining area) for reproduction or early life stages. It is stressed that there is no evidence that this is the case or that it is not. These groups have higher-than-average trophic importances, implying a higher potential for indirect (trophically-mediated) effects to arise from changes to these groups.

Table. 8.1. Potential effects of mining on productive capacity of groups with higher than average trophic importance based on (a) direct effect on spawning/early life stages; and/or (b) habitat-mediated effect. The revised rank trophic importances are taken from the medians shown in Fig. 8.3. Groups are colour coded: green=direct effect of mining likely to be low/negligible; amber=direct effect of mining may be moderate (at least for some species in group); red=direct effect of mining not known but potentially higher (at least for some species in group).

Rank trophic importance	Group	Location of spawning/early life stages. Habitat dependence.	Likely direct effects of mining/plume on productive capacity
1	Phytoplankton	Whole Chatham Rise; planktonic.	No significant impact.
2	Detritus benthic	Not relevant – no spawning	Negligible impact on in/out flows of detritus at scale of Chatham Rise.
3	Detritus water	Not relevant – no spawning	Negligible impact on in/out flows of detritus at scale of Chatham Rise.
4	Mesozooplankton	Can reproduce in water column or on/near the seabed. Planktonic.	Probably low overall.
5	Small demersal fish	Not known – diverse group. Likely to vary between species. Some may require hard benthic substrate for spawning/early life stages.	Not known, but could be high if key species in group spawn in/close to mining area.
6	Hoki	Spawn outside Chatham Rise area. Early life stages widespread in area.	Low direct impact.
7	Het. flagellates	Whole Chatham Rise; planktonic.	Negligible impact at scale of Chatham Rise.
8	Arthropods (e.g. prawns & shrimps)	Unlikely to move large distances. May depend on habitat (hard or soft) to spawn. Early life stages planktonic.	Probably low.
9	Meiobenthos	Whole Chatham Rise; very small scale movement.	Negligible impact at scale of Chatham Rise.
10	Bacteria_water	Whole Chatham Rise; planktonic.	Negligible impact on bacteria at scale of Chatham Rise.
11	Mesopelagic fish	All likely to be pelagic spawners. Early life stages planktonic.	Probably low, but not well known for most species in group.
12	Macrozoo krill	May be pelagic or benthic spawners. May require benthic habitat. Semi-nektonic.	Not known, but could be high if key species in group spawn in/close to mining area.

Table. 8.1. (continued)

13	Macrozoo gelatinous	Largely/entirely planktonic and likely to be pelagic spawners. Early life stages planktonic.	Probably low.
14	Cephalopods	Poorly known. Likely to vary between species and some may require had benthic habitat for spawning/early life-stages.	Not known, but could be high if key species in group spawn in/close to mining area.
15	Macrobenthos	Whole Chatham Rise. Small scale movement. Spawning will depend on suitable habitat (hard or soft).	Likely low at scale of Chatham Rise for group as a whole, but some species in group may occur only close to mining areas
16	Rattails & ghost sharks	Ecology not well known. Could make spawning migrations. Spawning could depend on hard benthic habitat.	Not known, but could be high if key species in group spawn in/close to mining area.
17	Hake guild	Hake spawning not near mining area. Spawning/juvenile areas of other species in guild less well known.	Probably low for hake. Not known for other species in guild.
18	Bacteria sediment	Whole Chatham Rise	Negligible impact on bacteria at scale of Chatham Rise.
19	Ling guild	Ling spawning not in mining area. Spawning/juvenile areas for other species in guild less well known.	Probably low for ling (though reasons for hotspot not known). Not known for other species in guild.

It is important to note that the analysis presented here provides information at one set of scales. The model provides information: (i) at the spatial scale of the whole Chatham Rise; (ii) averaged over an annual period (seasonal dynamics not resolved); (iii) for a "typical" recent year (i.e. between-year variations are not considered); (iv) in a relatively small number of trophic groups (intra-population demographics and particular species not resolved); (v) focus on major flows of energy through the food-web and so little information is provided with regard to minor species; (vi) trophic connections only (provision of habitat, predation-interference effects are not considered).

There is a large amount of information available on the food-web of the Chatham Rise; the Chatham Rise is probably the best-studied offshore region within the New Zealand EEZ. Nevertheless, substantial deficits in information remain in all groups. Particularly poorly known groups include cetaceans (numbers of whales in the study area at different times of the year are not known), mesopelagic fishes, and large zooplankton (both gelatinous and hard-bodied macrozooplankton).

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ANNEX 6.1 ATLANTIS functional groups

DFH - Dem fish shelf crust feed	Scorpaena elongata	Diplodus vulgaris	Caelorhynchus caelorhynchus
Argentina sphyraena	Trigla lyra	Epinephelus aeneus	Capros aper
Arnoglossus imperialis	DSH - Dem selaceens shelf	Hippocampus hippocampus	Carapus acus
Arnoglossus laterna	Dasyatis pastinaca	Labrus mixtus	Ceratoscopelus maderensis
Arnoglossus rueppelli	Mustelus asterias	Liza aurata	Chlorophthalmus agassizi
Arnoglossus thori	Mustelus mustelus	Mugil cephalus	Diaphus holti
Aspitrigla cuculus	Mustelus punctulatus	Muraena helena	Diaphus metopoclampus
Blennius ocellaris	Myliobatis aquila	Pagrus caeruleostictus	Diaphus rafinesquei
Bothus podas	Raja alba	Pagrus pagrus	Diaphus spp
Buglossidium luteum	Raja asterias	Phycis phycis	Electrona rissoi
Callionymus lyra	Raja batis	Sciaena umbra	Epigonus denticulatus
Callionymus maculatus	Raja brachyura	Scorpaena loppei	Epigonus telescopus
Callionymus risso	Raja circularis	Scorpaena notata	Evermanella balbo
Cepola macrophtalma	Raja clavata	Scorpaena porcus	Facciolella oxyrhyncha
Chelidonichthys gurnardus	Raja fullonica	Scorpaena scrofa	Gadella maraldi
Chelidonichthys lastoviza	Raja melitensis	Scorpaena spp	Gadiculus argenteus
Chelidonichthys lucerna	Raja miraletus	Spondyliosoma cantharus	Hoplostethus mediterraneus
Chelidonichthys obscurus	Raja montagui	Umbrina canariensis	Hygophum benoiti
Citharus linguatula	Raja naevus	DSS - Dem selaceens slope	Hymenocephalus italicus
Coris julis	Raja oxyrinchus	Centrophorus granulosus	Lampanyctus crocodilus
Dactylopterus volitans	Raja polystigma	Centrophorus uyato	Lampanyctus pusillus
Dalophis imberbis	Raja radula	Chimaera monstrosa	Lappanella fasciata
Deltentosteus quadrimaculatus	Raja spp	Dalatias licha	Lepidopus caudatus
Echelus myrus	Torpedo marmorata	Etmopterus spinax	Lestidiops jayakari jayakari
Gaidropsarus biscayensis	Torpedo nobiliana	Galeus melastomus	Lobianchia dofleini
Gaidropsaurus mediterraneus	Torpedo torpedo	Heptranchias perlo	Macroramphousus gracilis
Gaidropsaurus spp	DSM - Dem fish shelf mixed	Hexanchus griseus	Macrorhamphosus scolopax
Gnathophis mistax	Altri Serranidi	Oxynotus centrina	Maurolicus muelleri
Gobius cobitis	Balistes capriscus	Squalus acanthias	Micromesistius poutassou
Gobius cruentatus	Hippocampus spp	Squalus blainvillei	Myctophidae spp

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Gobius niger	Lithognathus mormyrus	ENG - Anchovy	Myctophum punctatum
Gobius spp	Mullus surmuletus	Engraulis encrasicholus	Nansenia oblita
FIGLepidotrigla cavillone	Serranus cabrilla	EPI - Epipelagic fish	Nemichthys scolopaceus
Lepidotrigla dieuzeidei	Solea solea	Anthias anthias	Nettastoma melanurum
Lesueurigobius friesii	Solea spp	Aphia minuta	Nezumia aequalis
Lesueurigobius sanzoi	Solea variegata	Boops boops	Nezumia sclerorhynchus
Lesueurigobius sueri	Sphoeroides pachygaster	Callanthias ruber	Notacanthus bonaparte
Microchirus ocellatus	Symbolophorus veranyi	Centracanthus cirrus	Notoscopelus elongatus
Microchirus variegatus	Symphodus mediterraneus	Glossanodon leioglossus	Paralepis c. coronogoides
Ophidium barbatum	Symphurus ligulatus	Scyliorhinus canicula	Paralepis HYA
Pagellus acarne	Symphurus nigrescens	Scyliorhinus stellaris	Sudis hyalina
Pagellus erythrinus	Symphurus spp	Spicara flexuosa	MSG - Mesop slope jelly feed
Pomatoschistus minutus	Syngnathus abaster	Spicara maena	Centrolophus niger
Psetta maxima	DSP - Dem shelf fish pisc	Spicara smaris	Cubiceps gracilis
Serranus hepatus	Conger conger	HKE - Hake	Ruvettus pretiosus
Synapturichthys kleinii	Lophius budegassa	Merluccius merluccius	Schedophilus medusofagus
Synchiropus phaeton	Lophius piscatorius	LPL - Large pelagics	MSP - Mesop slope fish pisc
Triglidae	Synodus saurus	Pteromylaeus bovinus	Chauliodus sloani
Trisopterus m. capelanus	Trachinus araneus	Xiphias gladius	Stomias boa
DFS - Dem fish slope	Trachinus draco	MPL – Medium pelagics	MUL – Mullus
Helicolenus d. dactylopterus	Trachinus radiatus	Naucrates ductor	Mullus barbatus
Lepidorhombus boscii	Uranoscopus scaber	Sphyraena sphyraena	SAR - Sardine
Lepidorhombus whiffjagonis	Zeus faber	Trachurus mediterraneus	Sardina pilchardus
Molva dipterygia	DSR - Dem fish shelf rocky	Trachurus picturatus	SB - Seabirds
Molva molva	Alepocephalus rostratus	Trachurus trachurus	SPL - Small pelagics
Mora moro	Dentex dentex	MSC - Mesop slope crust feed	Alosa fallax
Ophisurus serpens	Dentex gibbosus	Argyropelecus hemigymnus	Sardinella aurita
Pagellus bogaraveo	Dentex macrophthalmus	Bathypterois mediterraneus	Scomber colias
Peristedion cataphractum	Diplodus annularis	Bellottia apoda	Scomber scombrus
Phycis blennoides	Diplodus puntazzo	Benthocometes robustus	Scomber spp

Table 1 - Vertebrates species aggregation into functional groups

PL - Large Phytoplankton	Diatoms
PS - Small Phytoplankton	Picophytoplankton: Synechoccus, Prochlorochoccus, picoeukariotes
DF - Dinoflagellates	Dinoflagellates
ZS - Small Zooplankton	Copepodites
ZM -Mesozooplankton	Copepods
ZL - Large Zooplankton	Krill and chaetognath
ZG - Gelatinous Zooplankton	Salps (pryosomes), coelenterates
PB - Pelagic Bacteria	Pelagic Bacteria
CEP - Pelagic cephalopods	Abralia veraniji, Alloteuthis media, Alloteuthis spp, Alloteuthis subulata, Ancistroteuthis lichteinsteini, Argonauta argo, Bathypolypus sponsalis, Heteroteuthis dispar, Histhioteuthis bonnellii, Histhioteuthis reversa, Histhioteuthis spp, Illex coindetii, Loligo forbesi, Loligo vulgaris, Ommastrephes bartramii, Onychoteuthis banksi, Todarodes sagittatus, Todaropsis eblanae
BB - Sediment bacteria	Aerobic and anaerobic bacteria
BC - Carnivoruos infauna	Polychaetes
MBS - Macroepibenthos Slope	Alpheus glaber, Anamathia rissoana, Anapagurus laevis, Crustacea, Dardanus arrosor, Goneplax rhomboides, Isopoda, Lepas anatifera, Macropodia longipes, Macropodia longirostris, Macropodia rostrata, Pagurus alatus, Parthenope macrochelos, Parthenope massena
MBH - Macroepibenthos Shelf	Other Crustaceans, Clibanarius erythropus, Dardanus calidus, Diogenes pugilator, Dromia personata, Ebalia deshayesi, Eriphia verrugosa, Eurynome aspera, Ilia nucleus, Inachus dorsettensis, Inachus parvirostris, Inachus spp, Inachus thoracicus, Latreillia elegans, Paguri-Anomura, Paguristes eremita, Pagurus cuanensis, Pagurus prideaux, Pagurus spp, Pilumnus hirtellus, Pinnotheres pisum, Pisa armata, Pisa nodipes, Scalpellum scalpellum
DNS - Natant Decapods Slope	Acanthephyra eximia, Acanthephyra purpurea, Aristeus antennatus, Chlorotocus crassicornis, Crangonidae, Gennadas elegans, Pasiphaea multidentata, Pasiphaea sivado, Plesionika acanthonotus, Plesionika antigai, Plesionika edwardsii, Plesionika gigliolii, Plesionika heterocarpus, Plesionika martia, Pontocaris cataphractus, Pontocaris lacazei, Processidae spp, Rissoides desmaresti, Rissoides pallidus, Sergestes robustus, Sergestes spp, Sicyonia carinata, Solenocera membranacea
DRH - Reptant Decapods Shelf	Galathea intermedia, Homarus gammarus, Liocarcinus corrugatus, Liocarcinus depurator, Maja crispata, Maja goltziana, Maja squinado, Maja verrucosa, Medorippe lanata, Palinurus elephas, Scyllarides latus, Squilla mantis
DPS - Reptant Decapods Slope	Bathynectes maravigna, Calappa granulata, Ethusa mascarone, Galathea dispersa, Geryon longipes, Homola barbata, Liocarcinus arcuatus, Macropipus tuberculatus, Monodaeus couchii, Munida intermedia, Munida iris, Munida spp, Nephrops norvegicus, Palinurus mauritanicus, Paromola cuvieri, Polycheles typhlops

BO - Meiobenthos	Mainly composed of nematodes
MB - Microphytobenthos	Mainly sediment diatoms
ARF - Red prawn	Aristaeomorpha foliacea
PWL - Pink prawn	Parapaeneus longirostris
CEB -Benthic cephalopods	Eledone cirrhosa, Eledone moschata, Neorossia caroli, Octopus defilippi, Octopus macropus, Octopus salutii, Octopus spp, Octopus vulgaris, Pteroctopus tetracirrhus, Rondeletiola minor, Rossia macrosoma, Scaeurgus unicirrhus, Sepia elegans, Sepia officinalis, Sepia orbignyana, Sepia spp, Sepietta oweniana, Sepietta spp, Sepiola affinis, Sepiola intermedia, Sepiola rondeleti, Sepiola spp, Sepiolinae
MA - Macroalgae	Vidalia etc.
SG - Seagrass	Posidonia oceanica
DL - Labile Detritus	
DR - Refractory Detritus	
DC - Carrion	
NUTRIENTS	Dissolved organic nitrogen, Ammonia, Nitrate, Silicate, Phosphorous

Table 2 - Invertebrates, plants, detritus and nutrients

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ANNEX 6.2 Data for scenarios

Parameters	BAU		Fmsy rose shrimp	
	Rose shrimp (DPS)	Hake (HKE)		
F (last years)	1.1	0.87	0.84 (Fmsy)	0.18 (Fmsy)
SSB (last year)	11.600	4602	11.600	4602
Recruitment (average last 3 years)	6 billion	218 million	6 billion	218 million
F-at-age	Age 0: 0.47 Age 1: 2.01 Age 2: 0.40 Age 3: 0.40	Age 0: 0.06 Age 1: 0.66 Age 2: 0.94 Age 3: 1.20 Age 4: 0.93 Age 5: 0.66 Age 6: 0.84 Age 7: 1.97	Scaled on Fmsy	Scaled on Fmsy
Stock numbers	Age 0: 7.15E+06 Age 1: 8.96E+05 Age 2: 2.58E+04 Age 3: 7.09E+01	Age 0 226541 Age 1 53275.98 Age 2 14184.82 Age 3 2830.112 Age 4 1055.218 Age 5 478.349 Age 6 279.111	As BAU to start simulation	As BAU to start simulation

		Age 7+ 350.7		
Spatial closures (only the current FRAs)	Fishing grounds< 50m /3 miles from the coastline	Fishing grounds< 50m /3 miles from the coastline	Full closure of FRAs	Full closure of FRAs
Days-at sea	60900	60900	60900*(Fmsy /Fcur)	60900*(Fmsy/ Fcur)
Gear selectivity			From -25% fishing mortality on the 0 group (all the vessels equipped with sorting grids separators)	From -14% fishing mortality on the 0 group (all the vessels equipped with sorting grids separators)
Trade marks, ecolabelling and certification	0 (not yet implemented)	0 (not yet implemented)	Trade marks: in Italy, two ways are available to obtain a certification: i) Ministero delle politiche agricole alimentari e forestali, MIPAA is entitled to provide a Protected Designation of Origin (D.O.P.) or Protected Geographical Indication (I.G.T.) certification; but, even if the costs are	

			relatively low, in fact, it needs about 1000.00 euro to achieve a certification, because of very slow Italian bureaucracy it needs a long time to certificate a given product. Ecolabelling: Friend of the Sea certification. The cost is 2000,00 euro per boat or 3.500,00 euro per three boats. Larger fishery vessel society should to pay about 20.000,00 euro for the certification. Certification: the price of MSC certification is about 150.000,00 euro and the time to obtain such certification is between 12

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			and 18 months.	
Regulate access to fishing grounds	No restrictions to access fishing grounds in international waters for vessels registered by GFCM. IUU fishing taking place.	No restrictions to access fishing grounds in international waters for vessels registered by GFCM. IUU fishing taking place.	A list of authorized fishing vessels define the fleet exploiting the stock in the area.	A list of authorized fishing vessels define the fleet exploiting the stock in the area.

SOCIO – ECONOMIC DATA ITALIAN TRAWL FLEET (Data from DCF 2004-2015)

	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
N. fishers										1713	1821	
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	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	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	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	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	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	48.64	62.69	60.39	43.49	44.18	53.20	61.71	58.77	49.95	49.81	46.62	56.05
Costs/landings (million €/t)	0.0047	0.0056	0.0065	0.0063	0.0062	0.0057	0.0059	0.0061	0.0053	0.0065	0.0058	0.0061
Gross profits HKE (million €)	3.86	2.28	1.72	1.53	1.37	2.46	2.16	1.58	2.98	0.70	1.39	0.66
Gross profits DPS (million €)	17.03	14.84	5.18	6.16	7.21	12.88	16.53	13.48	17.75	10.80	15.80	18.28
Gross profits HKE/ ton	€ 1,977.867	€ 1,322.817	€ 1,076.972	€ 958.918	€ 998.666	€ 1,591.044	€ 1,421.251	€ 1,246.277	€ 2,139.111	€ 452.886	€ 1,004.858	€ 467.344
Gross profits DPS/ vessel	€ 479.14	€ 407.05	€ 133.14	€ 177.03	€ 201.06	€ 362.71	€ 447.52	€ 429.35	€ 584.06	€ 373.78	€ 546.13	€ 635.60

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ANNEX 6.3 GADGET models parameters

Fixed and estimated parameter values from both the single- and multi-species GADGET for hake, rose shrimp and horse mackerel.

			Single-species model			
Species	Component	Parameter	Fixed	Estimated	Fixed	Estimated
hake	Length-weight	a b	4.81e-06 3.116	-	4.81e-06 3.116	-
rose shrimp	Length-weight	a b	2.11e-06 2.589	-	2.11e-06 2.589	-
horse mackerel	Length-weight	a b	1.23e-05 2.876	- -	1.23e-05 2.876	- -
hake	Growth: lengthvbsimple function	L_inf hake.k	100 -	- 0.12	100 -	- 0.13

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rose shrimp	Growth: lengthvbsimple function	L_inf pape.k	44.6 mm -	- 0.36	44.6 mm -	- 0.33
horse mackerel	Growth: lengthvbsimple function	L_inf trac.k	44 -	- 0.23	44 -	- 0.22
hake	binomial	hake.beta	-	1.99	-	0.88
rose shrimp	binomial	pape.beta	-	5.15	-	7.7
horse mackerel	binomial	trac.beta	-	1.81	-	1.05e-06
hake	Suitability: user-defined double logistic function with constant right tail	hake.alBMT	-	0.77	0.77	-
		hake.arBMT	-	0.7	0.7	-
		hake.l50BMT	-	15.5	15.5	-
		hake.r50BMT	-	35	35	-
		hake.pBMT	-	0.09	0.09	-
		hake.alDP	-	2.7	2.7	-
		hake.arDP	-	3	3	-
		hake.l50DP	-	25.3	25.3	-
		hake.r50DP	-	70.8	70.8	-
		hake.pDP	-	0.008	0.008	-
		hake.alsurBMT	-	1.8	1.8	-
		hake.arsurBMT	-	0.6	0.6	-
		hake.l50surBMT	-	7.4	7.4	-
		hake.r50surBMT	-	29.4	29.4	-
		hake.psurBMT	-	0.16	0.16	-
		hake.TalBMT	-	0.8	0.8	-
		hake.TarBMT	-	0.8	0.8	-
		hake.Tl50BMT	-	20.6	20.6	-

		hake.Tr50BMT	-	33	33	-
		hake.TpBMT	-	0.07	0.07	-
rose shrimp	Suitability: exponential 50%	pape.acom	-	0.62	0.62	-
		pape.L50com	-	19.1	19.1	-
		pape.asur	-	0.43	0.43	-
		pape.L50sur	-	18.8	18.8	-
		pape.Tacom	-	0.54	0.54	-
		pape.TL50com	-	22.3	22.3	-
Horse mackerel	Suitability: exponential 50%	trac.acom	-	1.18	1.18	-
		trac.L50com	-	10.8	10.8	-
		trac.asur	-	1.73	1.73	-
		trac.L50sur	-	18.2	18.2	-
		trac.acomDP	-	1	1	-
		trac.L50comDP	-	4.9	4.9	-
hake	Natural mortality vector-at-age	Age 0	1.60	-	1.60	-
		Age 1	0.57	-	0.57	-
		Age 2	0.33	-	0.33	-
		Age 3	0.25	-	0.25	-
		Age 4	0.21	-	0.21	-
		Age 5	0.19	-	0.19	-
		Age 6	0.17	-	0.17	-
		Age 7+	0.16	-	0.16	-
rose shrimp	Natural mortality vector-at-age	Age 0	1.60	-	1.60	-
		Age 1	0.8	-	0.8	-
		Age 2	0.7	-	0.7	-

		Age 3	0.6	-	0.6	-
Horse mackerel	Natural mortality vector-at-age	Age 0	1.98	-	1.98	-
		Age 1	0.57	-	0.57	-
		Age 2	0.34	-	0.34	-
		Age 3	0.26	-	0.26	-
		Age 4	0.22	-	0.22	-
		Age 5	0.20	-	0.20	-
		Age 6	0.18	-	0.18	-
hake	Initial population (x 1e7)	Age 0	-	15	-	12.85
		Age 1	-	0.01	-	2.95
		Age 2	-	0.19	-	0.61
		Age 3	-	0.91	-	2.38
		Age 4	-	1.36	-	7.08
		Age 5	-	1.38	-	3.84
		Age 6	-	2.08	-	0.31
		Age 7+	-	3.75	-	1.33
hake	Initial population mean length at age	Age 0	16.4	-	16.4	-
		Age 1	20.9	-	20.9	-
		Age 2	26	-	26	-
		Age 3	31.6	-	31.6	-
		Age 4	38.1	-	38.1	-
		Age 5	46.2	-	46.2	-
		Age 6	53.9	-	53.9	-
		Age 7+	60.7	-	60.7	-

hake	Initial population standard deviation at age	Age 0 Age 1 Age 2 Age 3 Age 4 Age 5 Age 6 Age 7+	2.7 3.1 2.9 3.9 5.4 5.8 6.8 8.27	- - - - - - - -	2.7 3.1 2.9 3.9 5.4 5.8 6.8 8.27	- - - - - - - -
Rose shrimp	Initial population (x 1e8)	pape.inage0 (proportional to exp M)	-	15	-	12.85
Rose shrimp	Initial population mean length at age	pape.recl0	-	15.32	-	22.37
Rose shrimp	Initial population standard deviation at age	pape.recsd0	-	6.88	-	2.83
Horse mackerel	Initial population (x 1e7)	trac.inage0 (proportional to exp M)	-	2.29	-	4.88
Horse mackerel	Initial population mean length at age	trac.recl0	-	25.45	-	27.85
Horse mackerel	Initial population standard deviation at age	trac.recsd0	-	3.13	-	7.15
hake	recruitment mean length	hake.recl alpha	- 4.8e-06 3.1	9.14 -	- 4.8e-06 3.1	7.98 -

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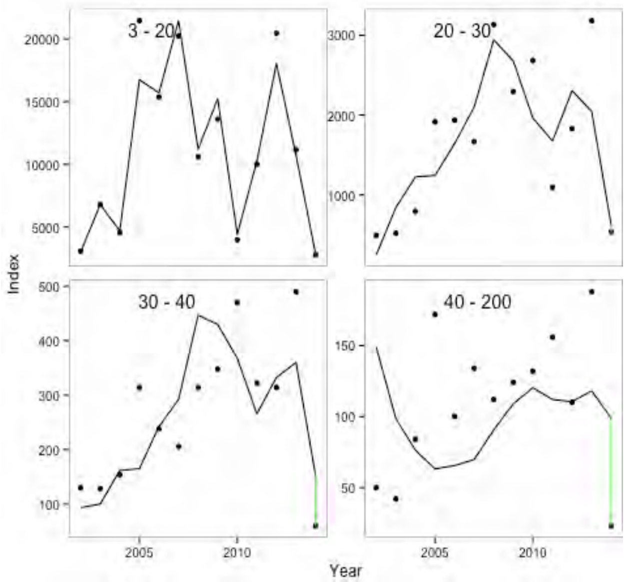
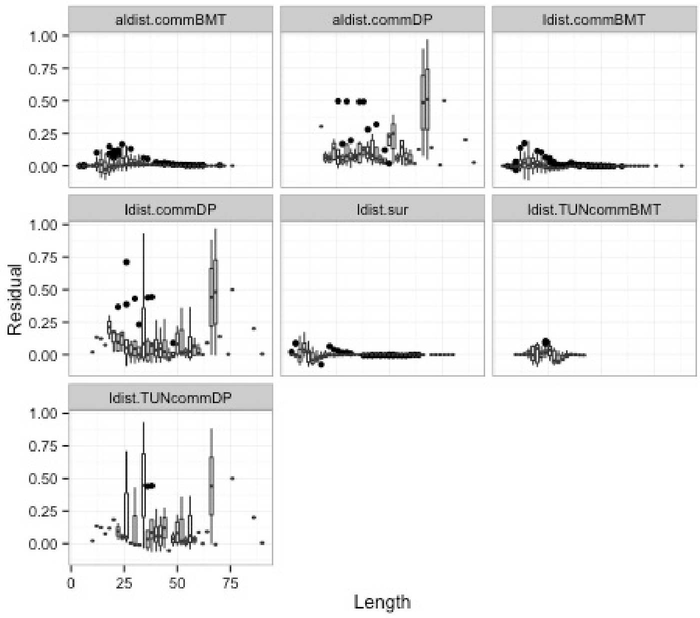
		beta		-		-
hake	recruitment std. deviation	hake.recsd		3.71		5.37
Rose shrimp	recruitment mean length	pape.recl alpha beta	- 2.11e-06 2.589	14.62 - -	- 2.11e-06 2.589	17.1 - -
Rose shrimp	recruitment std. deviation	pape.recsd	-	0.55	-	0.96
Horse mackerel	recruitment mean length	trac.recl alpha beta	- 1.23e-05 2.876	8.41 - -	- 1.23e-05 2.876	8.83 - -
Horse mackerel	recruitment std. deviation	trac.recsd	-	1.62	-	3.12

ANNEX 6.4 GADGET: fitting and residuals of singles and multispecies models

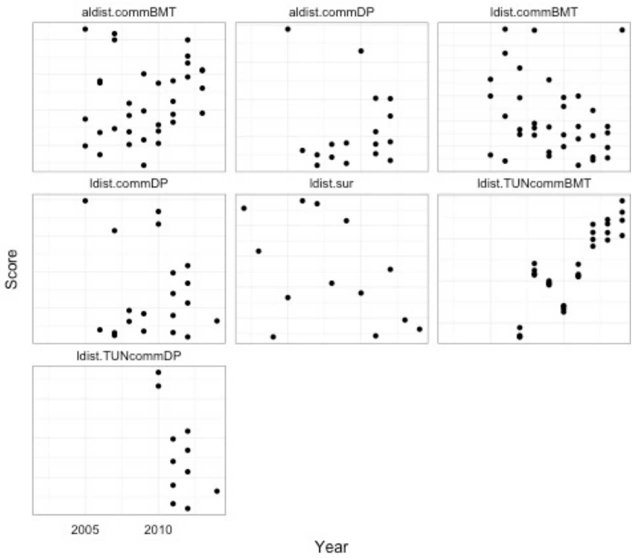
Single species GADGET model for hake (HKE)



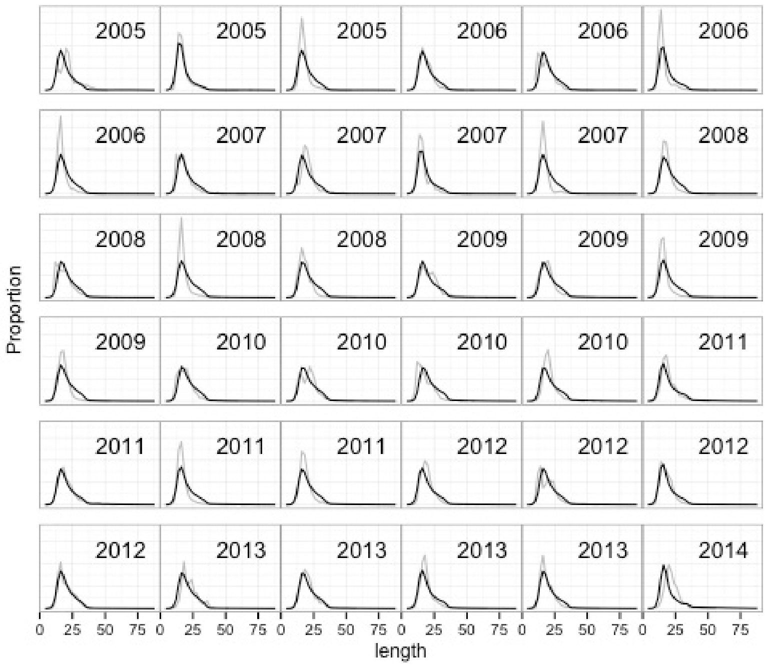
Single species GADGET model for hake: observed (black dots) and estimated (solid line) abundance indices from the MEDITS trawl survey



Single species GADGET Model Residuals for hake.



Single species GADGET Model likelihood scores for hake.



Single species GADGET model for hake: observed (grey line) and estimated (black line) annual length distributions of the Italian trawl fleet catches

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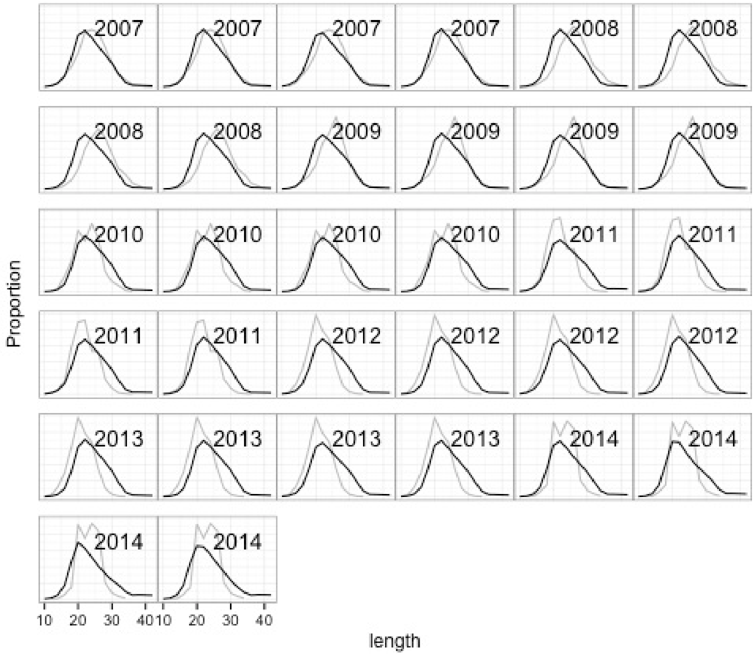
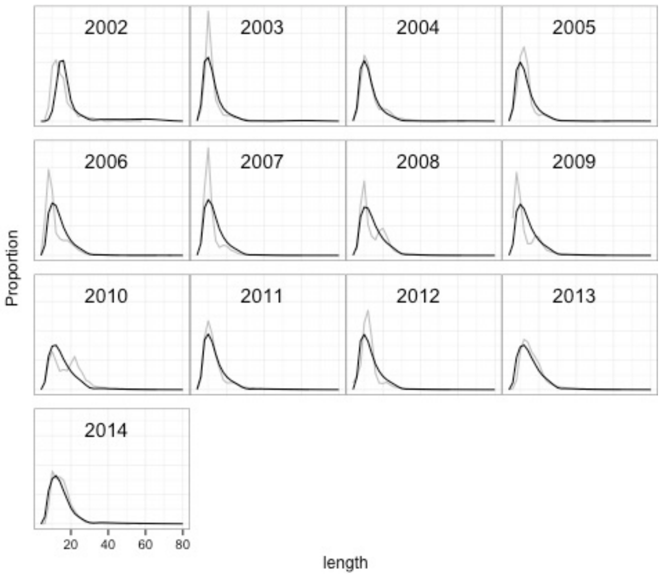
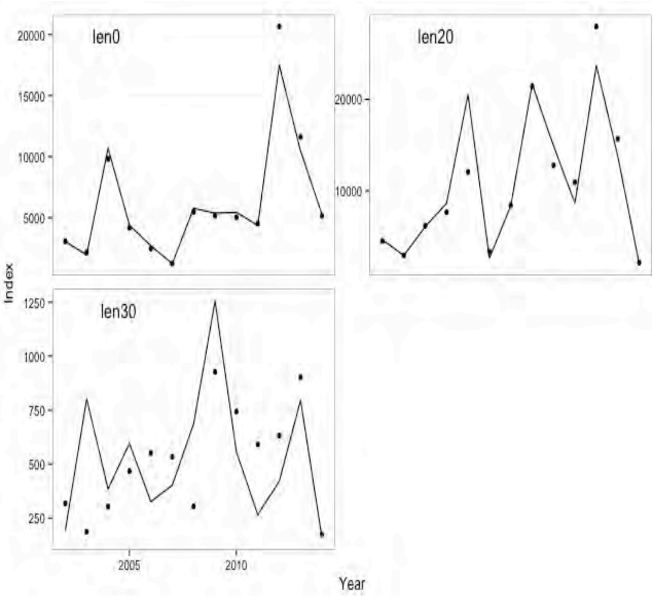


Single species GADGET model for hake: observed (grey line) and estimated (black line) annual length distributions of the Tunisian trawl fleet catches

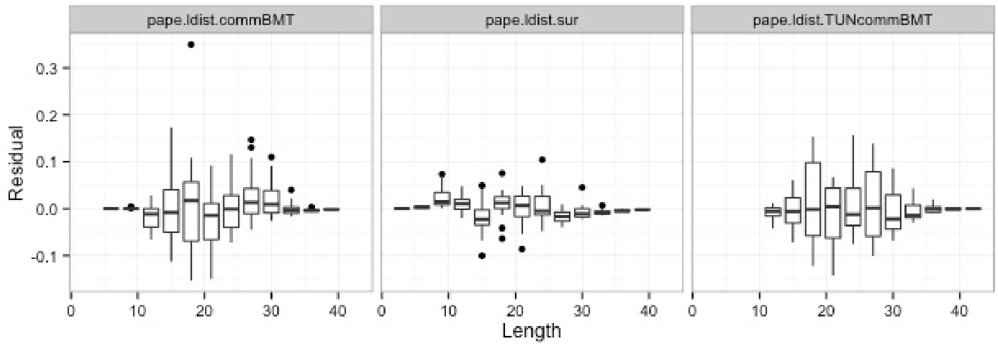


Single species GADGET model for hake: observed (grey line) and estimated (black line) annual length distribution of the MEDITS trawl survey.

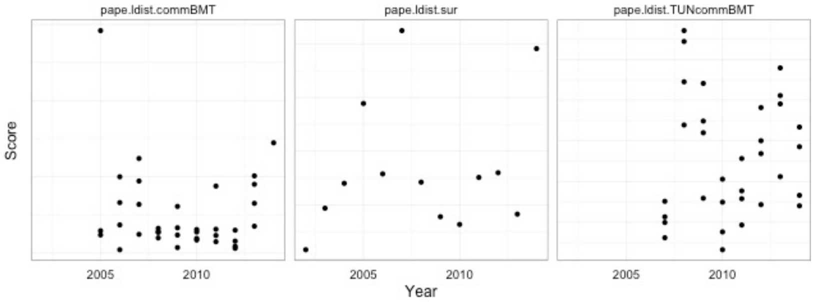
Single species GADGET model for deep water rose shrimp (DPS)



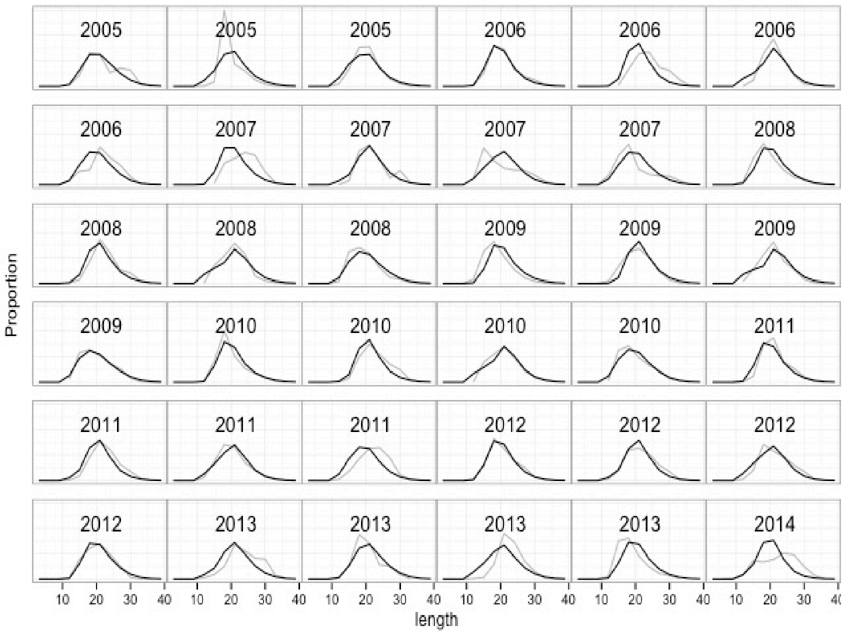
Single species GADGET model for rose shrimp (DPS): observed (black dots) and estimated (solid line) abundance indices from the MEDITS trawl survey



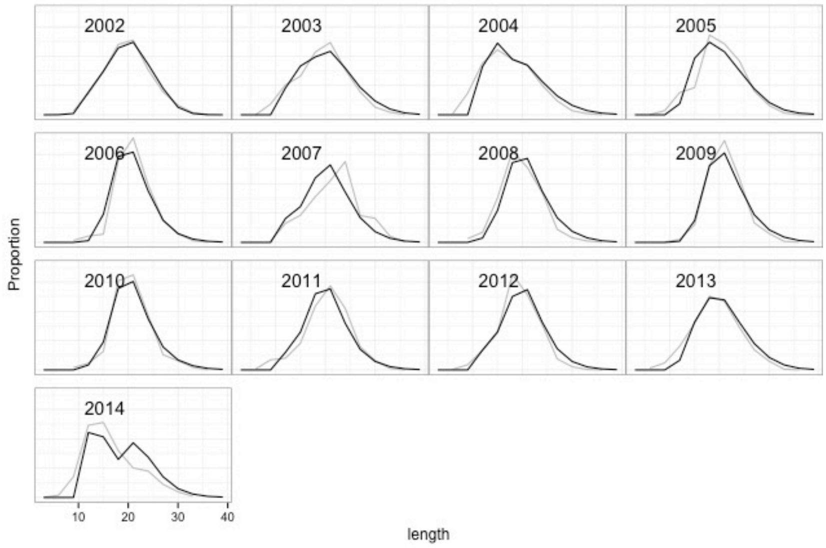
Single species GADGET Model Residuals for DPS.



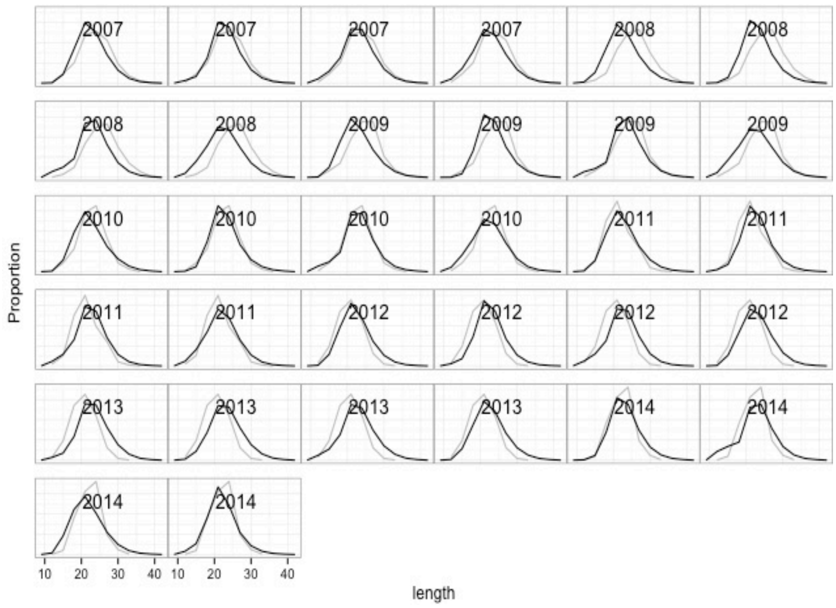
Single species GADGET Model likelihood scores for DPS.



Single species GADGET model for DPS: observed (grey line) and estimated (black line) annual length distributions of the Italian trawl fleet catches

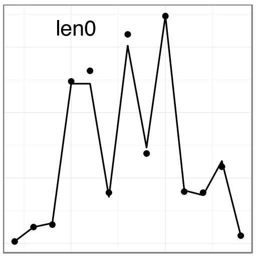


Single species GADGET model for DPS: observed (grey line) and estimated (black line) annual length distribution of the MEDITS trawl survey.

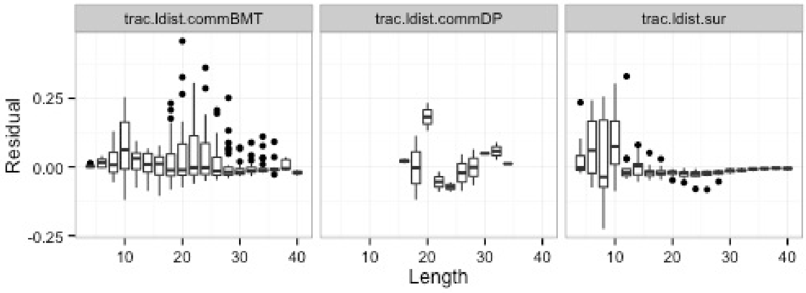


Single species GADGET model for DPS: observed (grey line) and estimated (black line) annual length distributions of the Tunisian trawl fleet catches

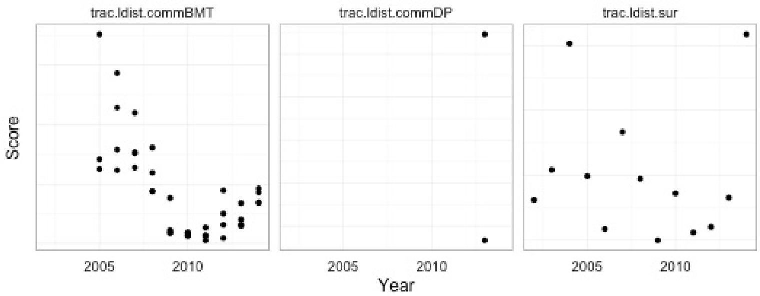
Single species GADGET model for horse mackerel (HOM)



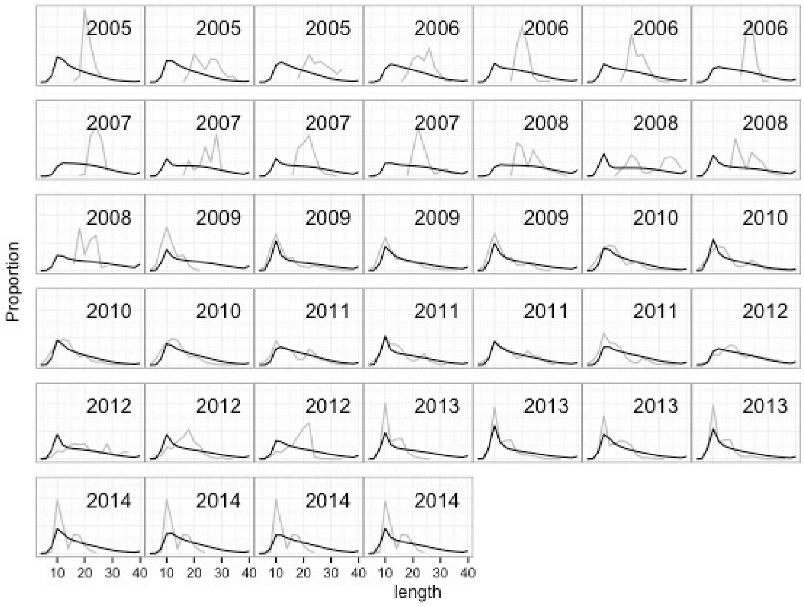
Single species GADGET model for horse mackerel: observed (black dots) and estimated (solid line) abundance indices from the MEDITS trawl survey



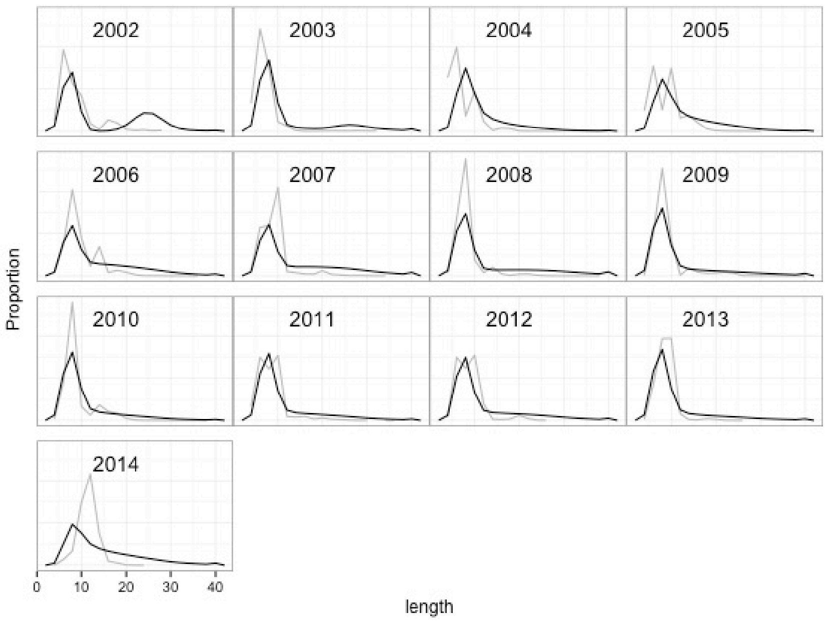
Single species GADGET Model Residuals for HOM.



Single species GADGET Model likelihood scores for HOM.

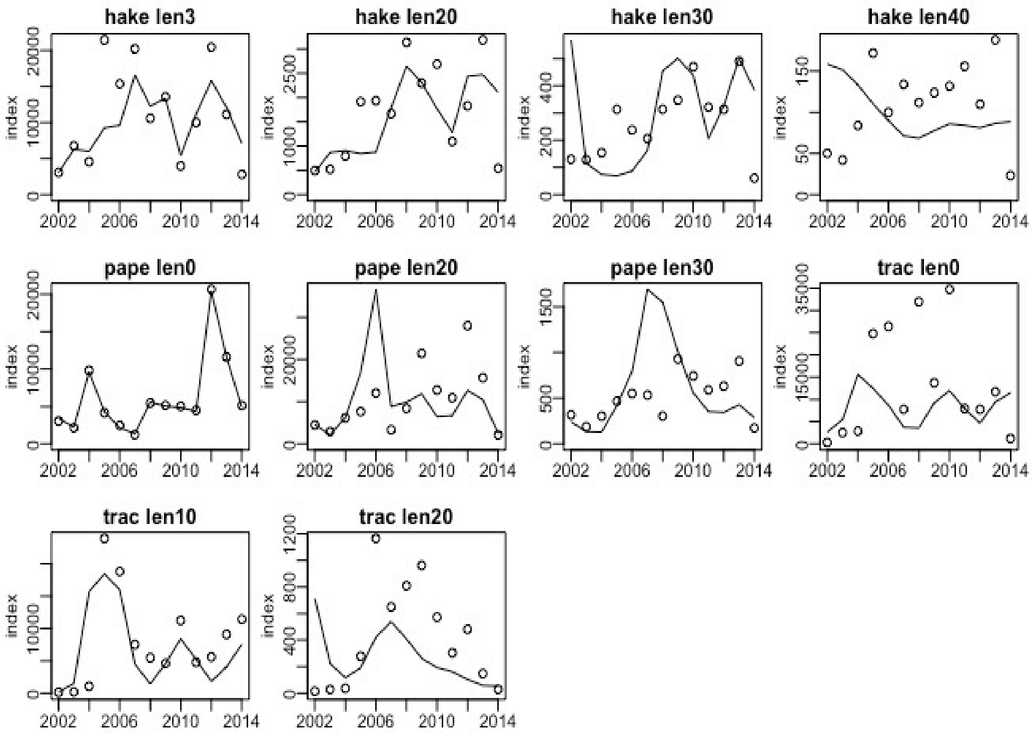


Single species GADGET model for HOM: observed (grey line) and estimated (black line) annual length distributions of the Italian trawl fleet catches

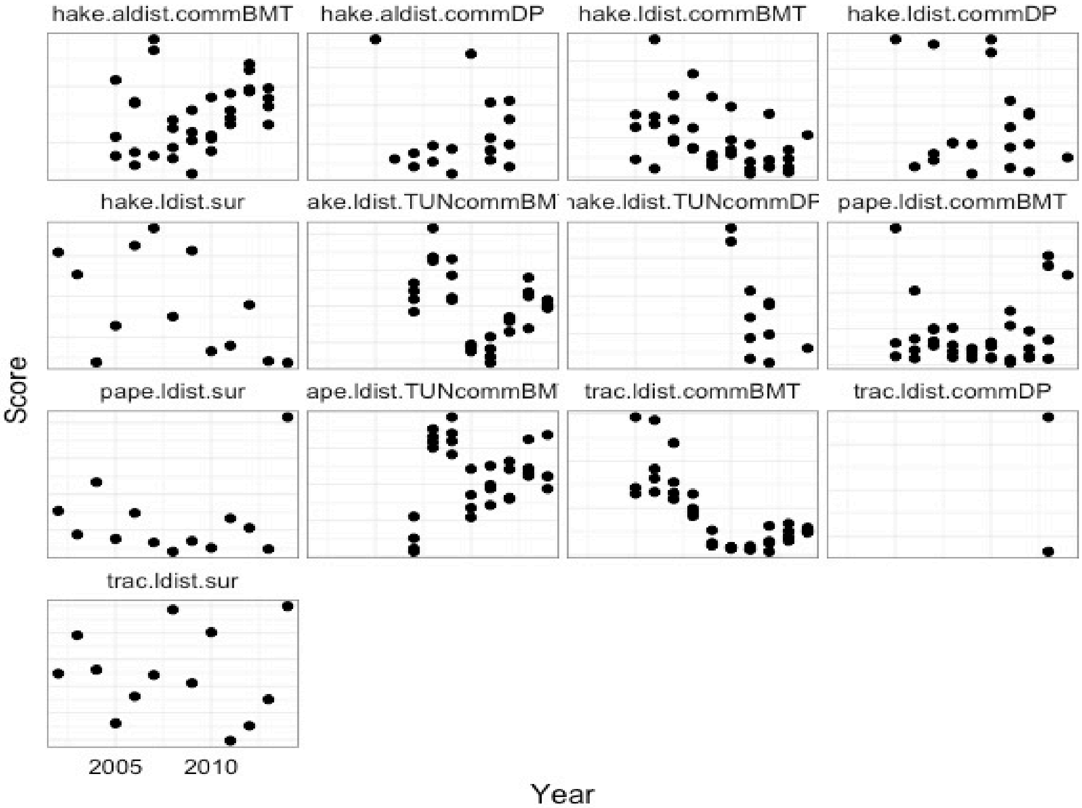


Single species GADGET model for HOM: observed (grey line) and estimated (black line) annual length distribution of the MEDITS trawl survey.

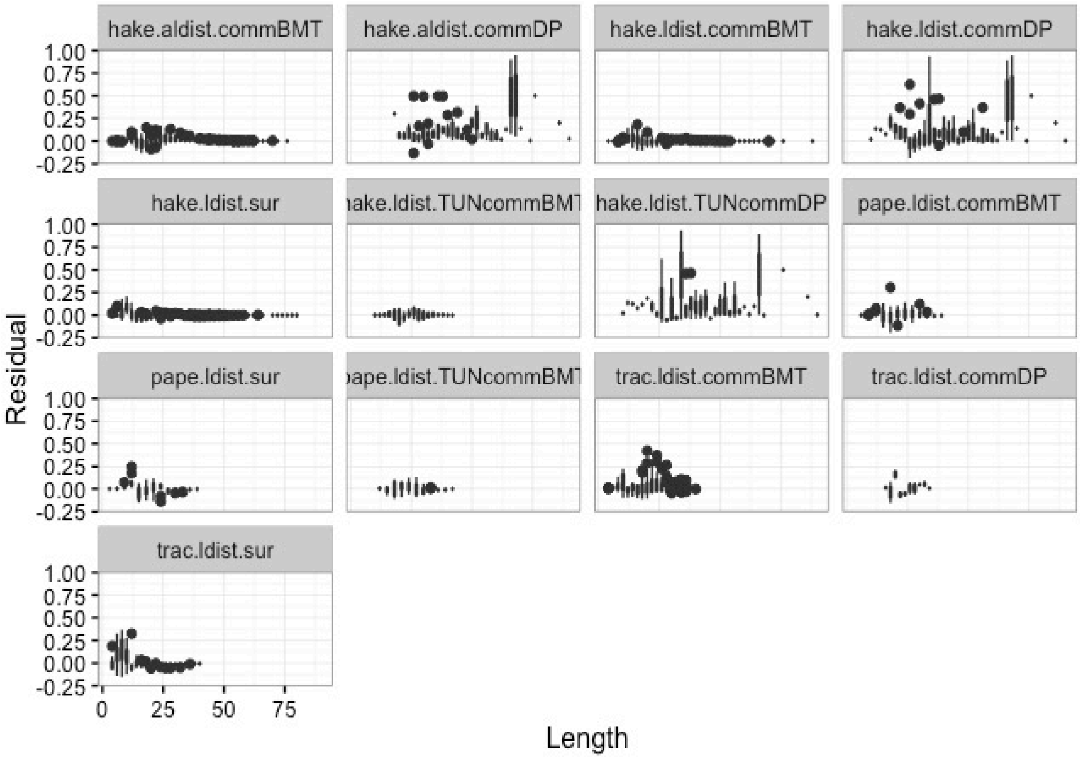
Multispecies GADGET model



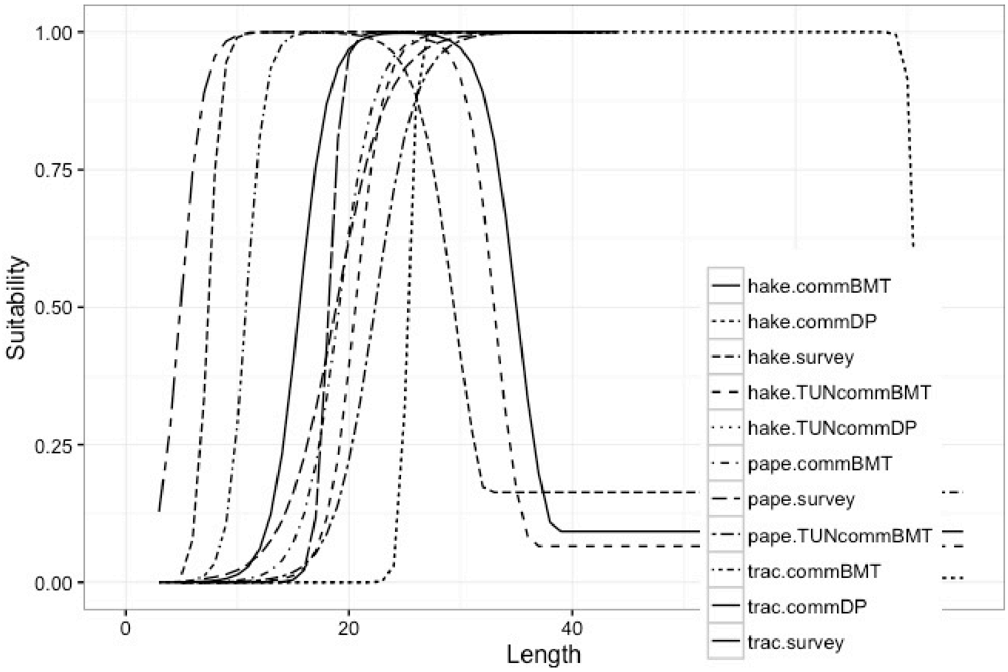
Multispecies GADGET Model: observed (dots) and fitted (solid lines) abundance indices of the MEDITS trawl survey



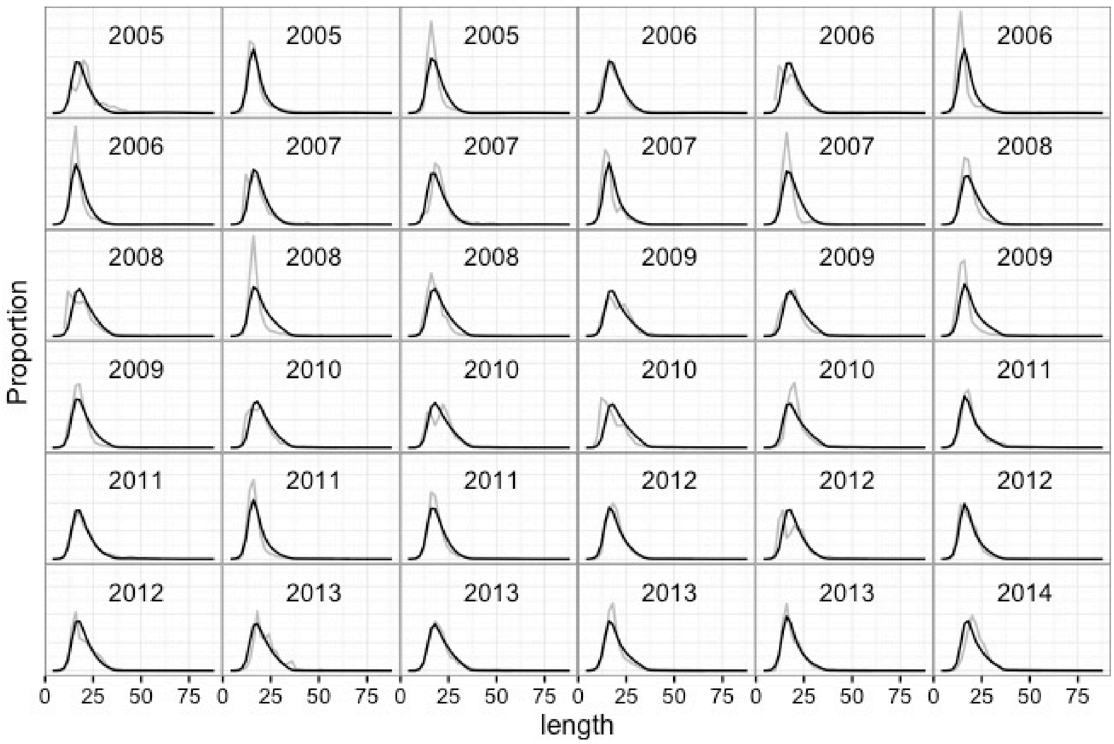
Multispecies GADGET Model Residuals likelihood scores by component and year



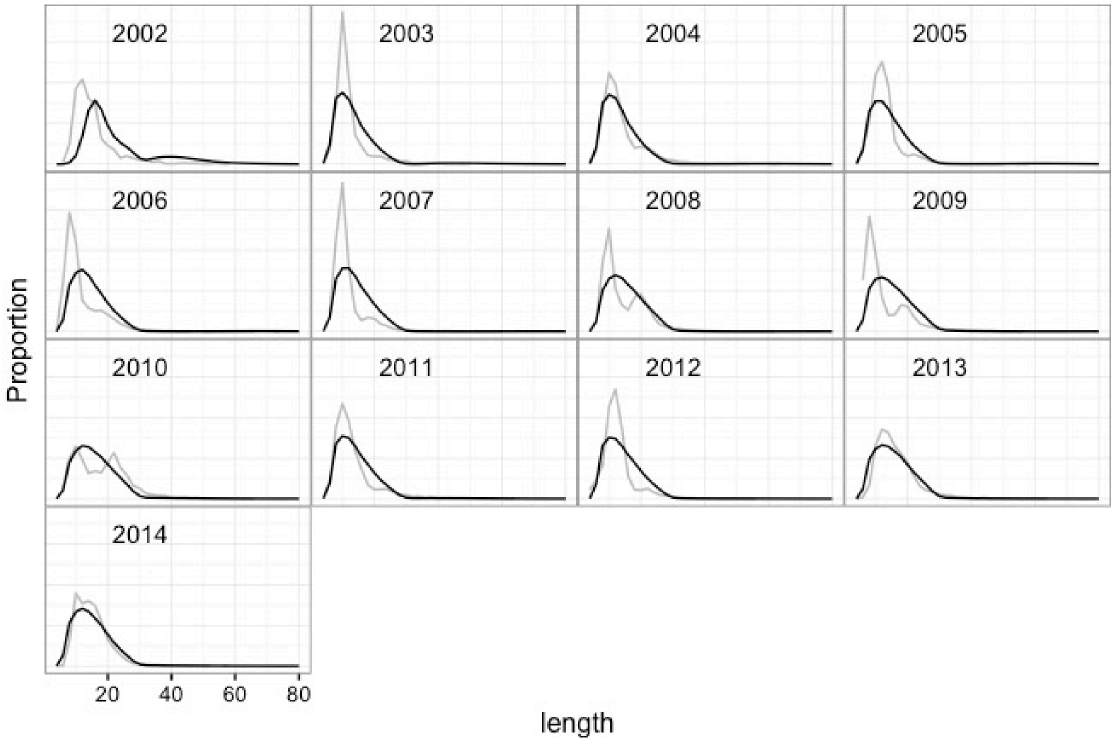
Multispecies GADGET Model Residuals



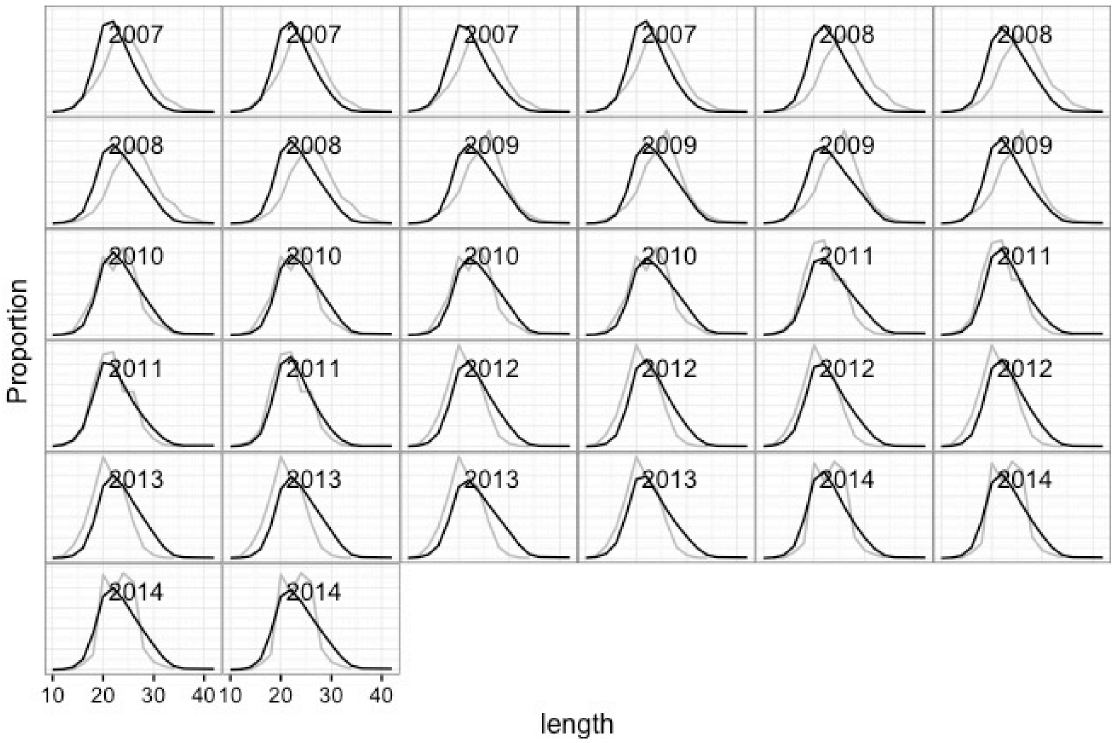
Multispecies GADGET Model: estimated fleets selectivity curves



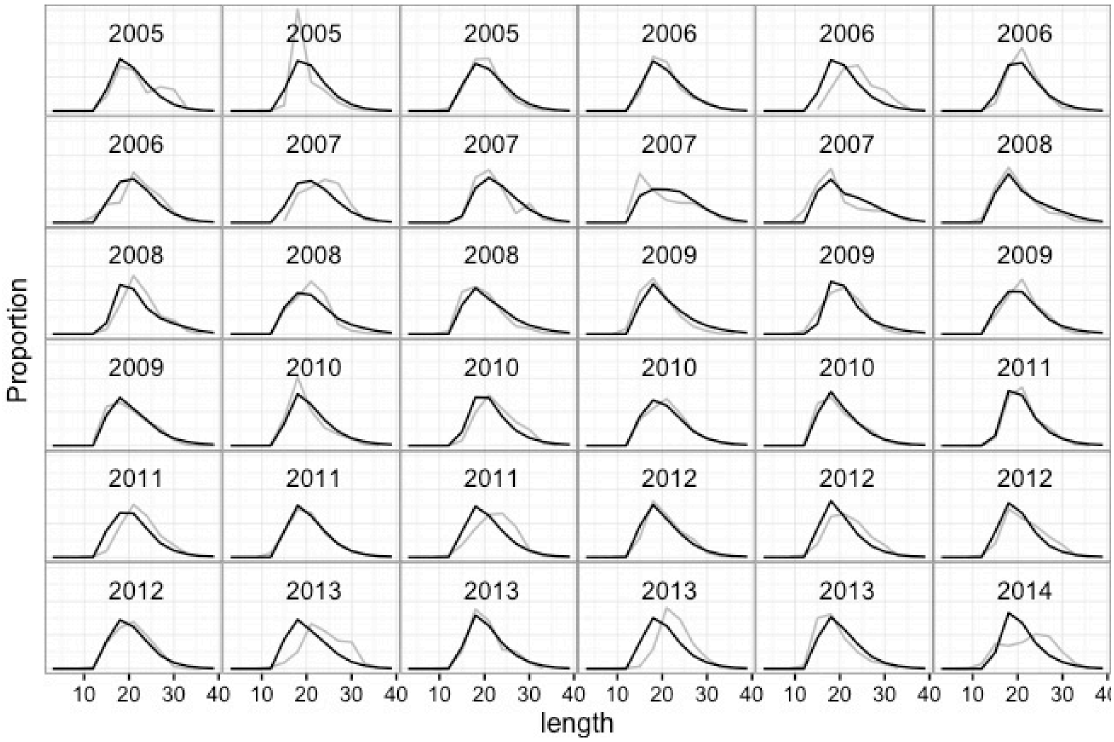
Multispecies GADGET Model: fitted (black line) and observed hake length distributions by year of the Italian trawl catch.



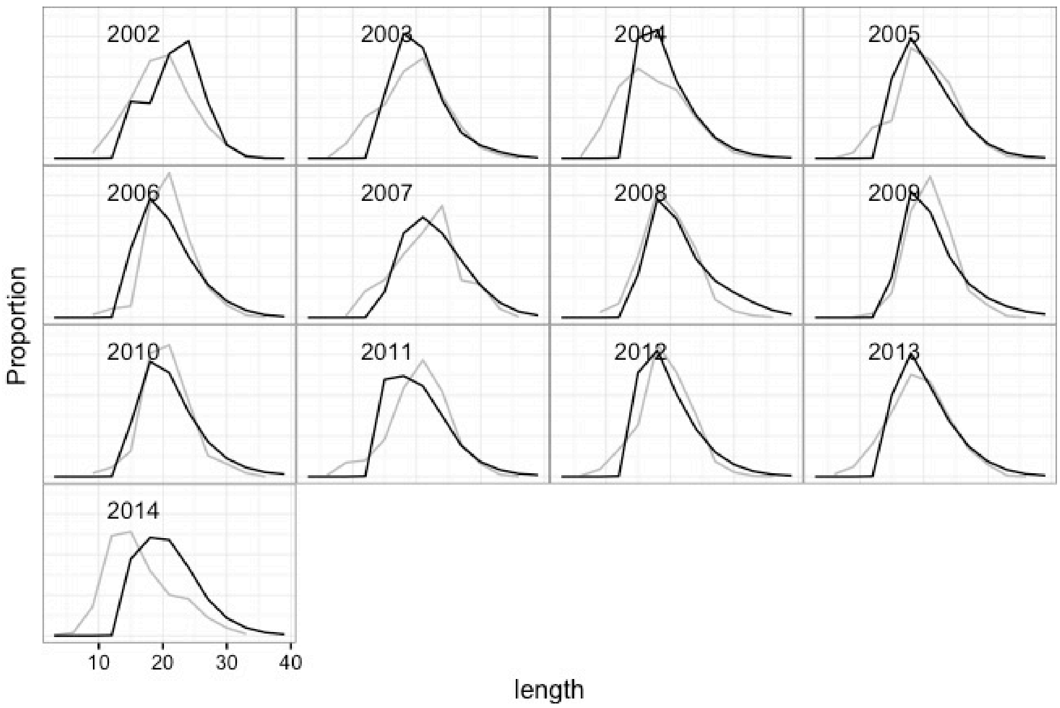
Multispecies GADGET Model: annual fitted (black line) and observed hake length distributions of the MEDITS trawl survey



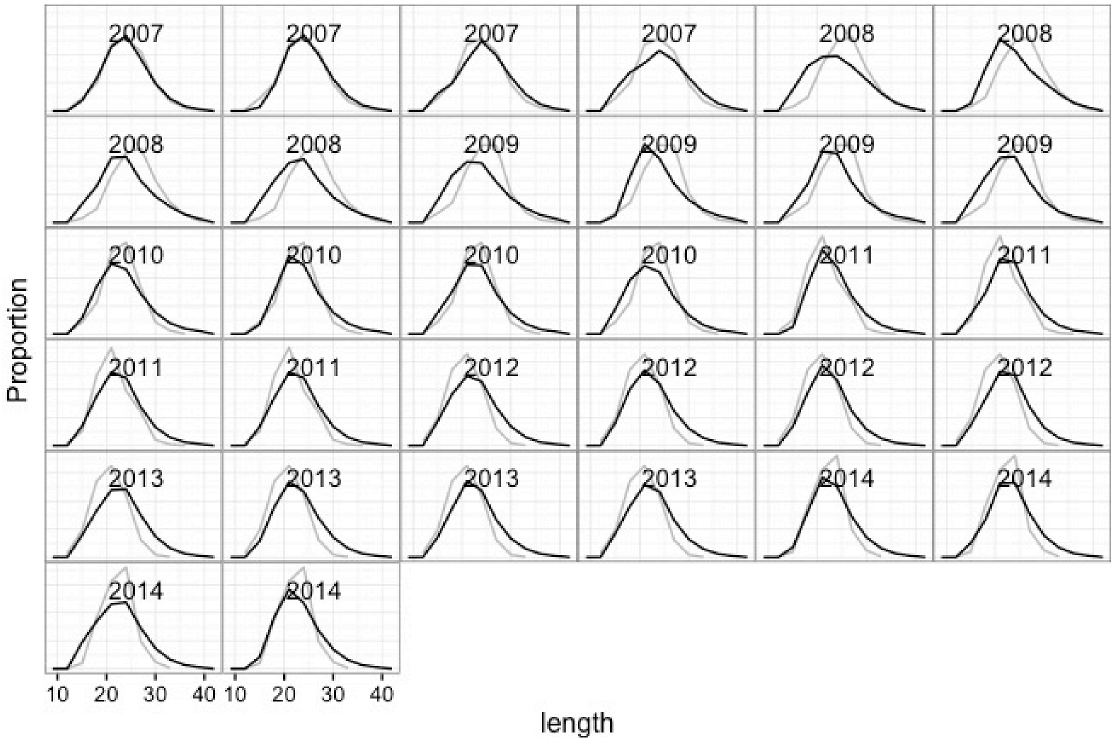
Multispecies GADGET Model: fitted (black line) and observed hake length distributions by year of the Tunisian trawl fleet catch.



Multispecies GADGET Model: fitted (black line) and observed DPS length distributions by year of the Italian trawl fleet catches.



Multispecies GADGET Model: annual fitted (black line) and observed DPS length distributions of the MEDITS trawl survey



Multispecies GADGET Model: fitted (black line) and observed rose shrimp length distributions by year of the catches of Tunisian trawlers.