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Sviluppo di un modello ecosistemico dell’Alto Adriatico

Development of an ecosystem model of the northern Adriatic Sea

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Abstract

The ecosystem approach to fisheries, encouraged since the 1980s', nowadays has still not been put in practice for an effective ecosystem based fishery management (EBFM). The use of ecosystem models in fishery science allows an integrated approach to study and quantify the potential effects of management scenarios or environmental changes on biological resources and fishery socioeconomics and can represent the core of an EBFM approach. The adoption of such ecosystem models for management is slow, and especially Ecopath with Ecosim and/or Atlantis have not yet become operational, have high development costs and are not always thoroughly validated. In this thesis, an ecosystem model was developed for the North East Adriatic Sea (NEAS), as a tool possibly used to support fisheries management in the area.

The developed ecosystem model integrates 33 functional groups, 10 fishing fleets and an aquaculture activity. The initial conditions and main parametrization are coherently embedded in a mass-balanced Ecopath model for the year 2005. The time-dynamic simulations in Ecosim for the period 2005-2015, was preliminary fitted to collected reference data and it were tested for sensitivity to the variation of initial parameters and forcing functions using Monte Carlo analysis. The results of the Monte Carlo analysis validated initial conditions and main parameters used for the model structure. Additionally, the largest variations in Ecosim simulations are produced by changes in production and consumption rate parameters. The sensitivity analysis indicated that fishing effort had a minor influence on the NEAS model, which was primarily influenced by primary productivity.

An ensemble of models was created to account for variability in food web interactions by implementing different fitting strategies for each model, without changing the mass-balance parameter settings or forcing functions. For each model in the ensemble, the vulnerability values were estimated by minimizing discrepancies between the predicted and observed data (reference time series). Although the area and the open boundaries represent model limitations, the model reproduced the observed dynamics of data by functional group with a good degree of accuracy.

The NEAS model was used to simulate the impacts of the latest Common Fishery Policy directive (2015-2030) and results used as a proxy for other areas of Mediterranean hosting multi-target multi-gear fisheries. A management scenario applying the EU Landing Obligation was simulated to assess the regulation's ecological and socioeconomic impacts. Additional scenarios aimed at reducing the Landing Obligation-subjected discards were also assessed and evaluated as alternative solutions for the discarding issue.

Results suggest that the combined direct and indirect effects of this regulation have globally a negative ecological and economic impact. The increase in landed biomass consequent of the regulation (approx. +13%) will cause an increase in fishermen workload, a small decrease in revenue (approx. -0.50%) and indirectly reduce biomasses at sea (approx. -0.20%) of both marketable and non-marketable species. Economic losses will not be compensated even with the possibility of selling landed discards for uses other than human consumption (e.g. fishmeal). Improving selectivity and introducing quotas resulted as the best alternatives, however none of the adaptation scenarios completely compensated the negative effects of the regulation. Results suggest the landing obligation is a management measure with several negative effects for systems characterized by mixed fisheries and the absence of output control (quotas), such as the Mediterranean Sea.

The careful model development and the analysis of different sources of uncertainties highlight the potentialities and limitations for its operational use in fishery management and the results of scenario analysis can support the ecosystem based management in the study area.

1. *Introduction*

1.1. Fishing and other impacts on the ecosystem

Fishing represents the main exploitation activity of wild resources for food and during the 20th century it had a major development and a transition to an industrial scale (Link, 2010; Piroddi et al., 2015; Fortibuoni, 2017). The primary goal of fishing technology throughout history was to develop better techniques to increase the quantity and diversity of landed fish (Link, 2010). This resulted in a race to fish and little attention was paid for decades to the impacts of such techniques on exploited stocks and their ecosystem (Kennelly and Broadhurst, 2002). A general consequence of decades of open access and rush for fish was the reduction in abundance of larger-sized fish species, usually the higher trophic levels, and successive targeting of smaller species such as small-sized demersal fish, small pelagic fish and other benthos representing the lower trophic levels (Pauly et al., 1998). These effects were summarized into the widely known "fishing down the food web" phenomenon (Pauly et al., 1998). During an inaugural address at the International Fisheries Exhibition in London in 1883, Thomas Huxley stated "*Probably all the great sea-fisheries are inexhaustible; that is to say that nothing we do seriously affects the number of fish ...*" (Smith, 1994) considering the limited capabilities to exploit the seas at the time. Over time, this quote has been proven wrong (Link, 2010) supporting Ray Lankester's quote, who at that same event stated "It is a mistake to suppose that the whole ocean is practically one vast store-house" (Smith, 1994). Nowadays, almost 90% of the world's major fisheries resources are fully exploited, overexploited or depleted resulting in a general decline in catches at global scale (FAO, 2016). The improvement of technology has allowed to exploit new fishing grounds, for example the deep seas (Morato et al., 2006) and the Antarctic (Ainley and Siniff, 2009; Nicol et al., 2012). In particular, the situation is very critical in the Mediterranean Sea, where large proportions of populations are overexploited and 85% of validated stock assessments indicate that fishing is going outside of biologically sustainable limits (GFCM, 2016a). The current fishing effort in the Mediterranean Sea is higher than target effort (F_{MSY}) in 90% of the stocks assessed in the period 2011-2014 by the EU's Scientific, Technical and Economic Committee for Fisheries (STECF) and FAO's General Fisheries Commission for the Mediterranean (GFCM) (see Cardinale and Scarcella 2017; Colloca et al., 2017).

The Mediterranean Sea is recognized for its high biodiversity (Bianchi and Morri, 2000), but at the same time it is the most impacted eco-region of the world (Coll et al., 2012), resulting in major alterations of the marine ecosystem and widespread conflicts among the users (Halpern et al., 2008). One consistent impacting factor on the Mediterranean ecosystem is the fishery and it does it in several ways: i) by heavily reducing resources at sea resulting even in local extinctions (e.g. Fortibuoni et al., 2010), ii) by changing the age- and size-structure of exploited populations (Colloca et al., 2013) and iii) by altering marine habitats (Pranovi et al., 2000). However, in the last centuries other human activities have also had a constantly increasing impact on the ecosystem through pollution, eutrophication, climate change, marine traffic, modification of marine habitats and diffusion of non-indigenous species (Cohrane et al., 2009; Galil, 2008; Mozetič et al., 2010; Coll et al., 2012; Micheli et al., 2013; Pećarević et al., 2013). Rebuilding the resources of exploited populations requires to combine each single species exploitation target (Maximum Sustainable Yield, MSY) with specific goals of the ecosystem approach to fisheries (EAF) in order to maintain community structure and the ecological role of each species (Walters et al., 2005; Cochrane and De Young, 2008; Colloca et al., 2013). There is evidence that in order to improve the biomass, yields or revenues of commercial stocks, it is not only necessary to reduce current fishing mortality to MSY reference values, but additional improvements such as changes in fishing selectivity are needed (Colloca et al., 2013). Marine spatial planning (MSP) might also help to reconcile the different uses of marine ecosystems for providing goods and services, including the provision of fisheries product (Micheli et al., 2013; Liquete et al., 2016). It is worth noting that restoring marine ecosystem structures and their resilience has potential to improve ecosystem services and redundancies: improvements of the reservoir of biodiversity and maintaining ecosystem functionality have the potential to be beneficial for fisheries too.

1.2. Usefulness of ecosystem models in fisheries management

1.2.1. Ecosystem approach to fisheries

The causes of the present overexploited and depleted natural resources should not be attributed only to the rush for fish and the technological improvements, i.e. the unchecked construction of bigger vessels combined with advances in electronic equipment and netting material and design (Kennelly and Broadhurst, 2002). Distinct Scientific communities were focusing on their own respective fields (ecology, oceanography and fisheries biology) with little integration since very recently. Since the late 1890s, clear emphasis has been placed on single species in fisheries science (Link, 2010). Only from the 1960's multispecies models have been developed and species interactions and the ubiquity of bycatch been recognized, but their use remained theoretical. From the 1980's the previously separated disciplines started to integrate with each other and fuse into ecological theories, such as the food web, the trophic cascade (Carpenter et al., 1985), but still were seldom used for ecosystem approach to fisheries. Even today fisheries management mostly rests on advice based on single species assessments, modelling population dynamics on "intraspecific" properties only. Often those models ignore the species interaction with the rest of the food web and its environment by assuming natural mortality to be constant and not influenced by changes in predator and prey abundances (Link, 2010). Ecosystem based fishery management (EBFM) has been proposed as a tool to be used for management advice, given its ability to account for a broader ecological context and interspecific interactions. Although EBFM potentials are high, institutional inertia, difficult comprehension, complexity of its application and lack of comprehensive data slowed down its adoption by the policymakers and stakeholders.

Although a more holistic approach to fisheries may seem to have taken place only recently, signs of awareness of species interconnections with the ecosystem were present even back in the 19th century. In 1873, Spencer Baird, the founder of the US Fisheries service, listed some possible causes for the decline in the New England fish stocks, among them: migration caused by reduction of prey, migration in search for food, diseases or environmental factors, excessive predation and finally the human impact, both due to pollution and overfishing. Baird’s hypothesis may be out-dated but its validity is undoubted to this day and is expressed in more modern terminology like ecosystem approach to fisheries (EAF) and ecosystem-based fishery management (EBFM). The EAF is more oriented to single species fisheries management while recognizing the effects of other sectors on the fisheries and vice versa and the effects of the fisheries on the ecosystem, e.g. the gear impact on the benthic habitat (Link, 2010). On the other hand, EBFM has a more holistic, synthetic and integrated approach in managing fisheries by coordinating, accounting for and including several factors, such as multispecies interactions, external drivers of productivity, fisheries discards. Although fisheries are still the main focus, this approach includes other organisms) from marine mammals and birds, to protected species, to non-target species) as well as environment-driven factors (e.g. primary production and chemical-physical properties of the water) that may influence the species behaviour and metabolism.

1.2.2. Ecosystem modelling

Marine ecosystems are communities of living organisms in conjunction with the non-living components of their environment interacting as a system (Tansley, 1935). Prey-predator relations and interactions with the environment are the base for elaborated food webs, responsible for the cycling and recycling of biomass and energy. Acknowledging the complexity of the ecosystem is the first step towards an ecosystem-based approach and opportune tools are required for that task. Ecosystem models developed so far represent the ecosystem as an organized structure and account quantitatively for the flow of energy and biomass within it. Through predation these models can also account for intraspecific dynamics such as cannibalism and interspecific interactions such as density dependent effects on food supply. Ecosystem models often represent, implicitly or explicitly, the energy and biomass dynamics of primary producers, which are strongly related to nutrient availability (mainly phosphorous and nitrogen) and favourable environmental conditions (usually temperature and solar radiation are fundamental drivers, but also water column mixing and shading can have important roles). Other organic material, usually metabolic by-products, faeces and carcases, represent an energy source for the so called detrital loop/cycle. Human activities may also be represented in ecosystem models, and in fisheries such models consider broader socioeconomic effects as well as ecosystem effects of anthropogenic (fishing) and environmental activities (Plagányi, 2007) (Table 1). Building complex models was limited by computational power in the past, but nowadays these are no longer a limiting factor as high performance computing (HPC) is becoming more common. Also, a constantly increasing quantity of ecological data is becoming available for integration into ecosystem models. However, our understanding of ecosystem functioning will still remain a limiting factor in developing more complex ecological models. An increase in model complexity to better account of biological realism can lead to associated increase in scientific uncertainty due to the limited knowledge about the parameters and the processes. Although the right choice of methods and model details depend on the ecological questions and the research objectives (Table 1), the evaluation of the research topic from different aspects by using a suite of different models may provide more complete explanations (Fulton and Smith, 2004). As suggested by Plagányi (2007), models could be divided in two categories: minimum realistic models (MRM) and whole ecosystem models (Table 1). The former are models that represent only a subset of the ecosystem in great detail, and can be focused either on the higher (Model of Intermediate Complexity for Ecosystem assessments (MICE), Plagányi et al., 2012) or lower trophic levels (Biogeochemical Flux Model (BFM), Lazzari et al., 2012) (Figure 1), leaving out of the model the description of other ecosystem components. On the other hand, whole-of-ecosystem models represent all trophic levels in an ecosystem, from primary producers to top predators, and can vary a lot in terms of structural resolution (number of state variables or ecosystem components represented). Some examples of these models are ATLANTIS (Fulton et al., 2011) and Ecopath with Ecosim (Christensen and Walters, 2004) (Figure 1). In terms of number of represented components, whole-of-ecosystem models typically describe more groups (for describing majority of species living in the ecosystem) than the MRMs (few key species represented). Whole-of-ecosystem models may be more appropriate for addressing broad-scale strategic (long-term) questions regarding the structure of the ecosystem, policy exploration and ecosystem-based fishery management. On the other hand, MRM models, aim to provide more specific tactical advices for management of focal species included in the model (e.g. species biomass or total allowable catches, TACs): these targets are defined at single species level but account for the main interactions with other species represented in MRM models, which therefore represent an accurately chosen subset of the ecosystem (Punt et al., 2016).

Table 1. Categorization of models according to feeding relationships, the primary model focus: effects of non-target species on commercial prey species, effects of fishing on the population of interest or on effects operating in both directions. Adaptation from Plagányi (2007). In “efficient predator” models it is assumed that the predator get its daily ration, while in “hungry models” predators are assumed to compete with predators of the same and other species for the vulnerable proportion of prey.

Model	Name	Model Units	“Efficient predator” or “Hungry predator”	Primary model focus
ATLANTIS	Atlantis	Nutrient	Hungry predator	Effects in both directions
Bioenergetic/ allometric model	Multi-species trophodynamic model using bioenergetics and allometric approach	Biomass	Both	Effects in both directions
BORMICON	BOReal MIgration and CONsumption model	Biomass	Both	Effects of fisheries on protected or other species
ERSEM II	Commission for the Conservation of Antarctic Marine Living Resources	Nutrient	Hungry predator	Effects in both directions
EwE	Ecopath with Ecosim	Biomass, Nutrient	Hungry predator/ Efficient predator	Effects in both directions
GADGET	Globally applicable Area Disaggregated General Ecosystem Toolbox (derived from BORMICON)	Biomass derived from population size structure	Both	Ecosystem effects on target population
IBM	Individual-based Models	Biomass	Efficient predator	Ecosystem effects on target population
IGBEM	Integrated Generic Bay Ecosystem Model	Nutrient	Hungry predator	Ecosystem effects on target population
INVITRO	INVITRO	Biomass	Efficient/Hungry predator	Effects in both directions
KPFM	Krill-Predatory-Fishery Model	Biomass	Efficient predator	Effects of fisheries on protected or other species
MICE	Model of Intermediate Complexity for Ecosystem assessments	Numbers at age, Biomass	Efficient predator	Dynamics of focal species and their predators or preys
MRM	Minimally Realistic Model	Biomass	Efficient predator	Ecosystem effects on target population
MSM	Multi-species Statistical Model	Biomass	Mixed	Limited effects in both directions
MSVPA and MSFOR	Multi-species Virtual population Analysis and multi-species Forecasting Model	Biomass	Efficient predator	Ecosystem effects on target population
MULTSPEC	Multi-species model for the Barents Sea	Biomass	Efficient predator	Ecosystem effects on target population
OSMOSE	Object-oriented Simulator of marine ecosystem Exploitation	Biomass at different levels of aggregation	Efficient predator but can starve	Effects in both directions
ESAM	Extended Single-species Assessment Models – models that are extensions to more conventional single-species stock assessment models	Biomass	Efficient predator	Ecosystem effects on target population
SEAPODYM	Spatial Ecosystem, and population Dynamics Model	Biomass	Efficient predator	Ecosystem effects on target population
SEASTAR	Stock Estimation with Adjustable Survey observation model and Tag-Return data	Biomass	Efficient predator	Ecosystem effects on target population
SMOM	Spatial Multi-species Operating Model	Biomass	Efficient predator	Ecosystem effects on target population
SSEM	Shallows Seas Ecological Model	Nutrient	Efficient predator	Ecosystem effects on target population

There is not one single best solution for modelling an ecosystem, even models designed with a specific purpose have their strengths and weaknesses. A way to walk around this problem is to use the strengths of each model in their respective field, and let other models take care of the weak points (Steenbeek et al., 2015). This model "cooperation" is known as model coupling, where models communicate with each other the relevant information for their simulations. In this way, complex physical, chemical and biological interactions can be integrated into a single system, the end-to-end modelling, spanning from physical circulation models through lower trophic level (LTL) biogeochemical models and higher trophic level (HTL) food web models (Akoglu et al., 2015). Such end-to-end ecosystem model can be virtually further expanded without limits by adding more specific models, for example: to account the human activities by use of fishermen behaviour (Wise et al., 2012), socioeconomic and other models.

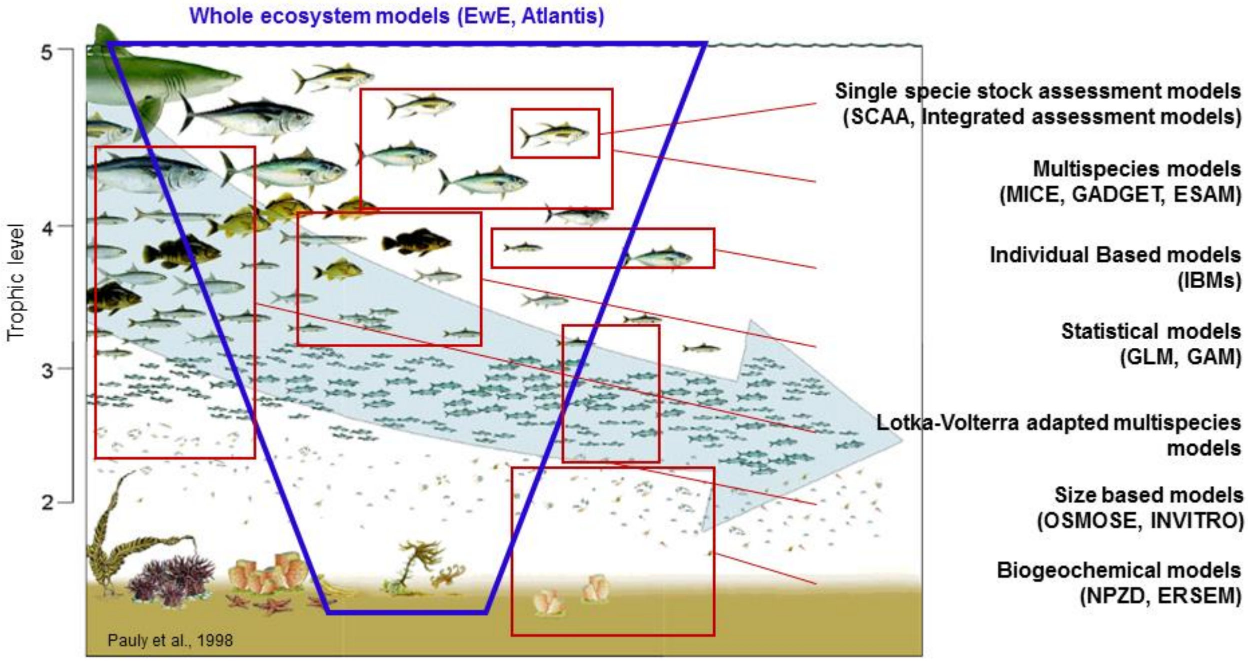

Figure 1. Schematic representation of various model domains of simulation. Modified after D. Pauly (1998).

Ecosystem models are not only about reconstructing and simulating ecosystems. The model development process involves, sometimes very tough and fierce, interaction between scientists, national experts, research organizations and various stakeholders to collect, compare and unify data in an ecologically sound way (e.g., JAKFISH project, Röckmann et al., 2012). Therefore, the "by-products" of the process also include the identification of data gaps and the definition of common goals between the collaborating parties (Cochrane and De Young, 2008). Although it may be a process were conflicts arise, it also represents an important learning process for all involved parties by raising and unifying the shared common knowledge to similar levels; a requirement for future development (Link et al., 2008). An especially important part of the process is enabling the creation of sense of ownership by the interested groups of the derived product, particularly required when policy goals have to be discussed for defining future management plans (Mion et al., 2015).

Stakeholder participation and dialogue need to be seen as integral components of multi-species fisheries management and scientists should avoid the temptation to use loosely constructed ecosystem models to justify a preferred point of view (Plagányi, 2007). Furthermore, it would be good practice if ecosystem models planned for use in management recommendations and decisions were subject to the same level of scrutiny as single-species models used for stock assessments. Otherwise there is a risk that considerable time and effort be wasted developing a model that will then be rejected when approaching management applications, or that bad management decisions based on poor scientific advice lead to potentially serious consequences and mistrust.

1.3. Application of ecosystem models in the Mediterranean Sea

An increase of public awareness demanding for better management of marine resources (Micheli et al., 2013b) and the development of ecosystem-based approaches (Sartor et al., 2014) have also been observed in the Mediterranean region. The adoption of more holistic ecosystem approaches to resource management is a basic requirement of international treaties and agreements, such as the Convention on Biological Diversity (CBD) or the UN Framework Convention on Climate Change (UNFCCC) (Coll and Libralato, 2012). Scientists around the Mediterranean therefore responded by increasing their research activities in ecosystem-based studies: ecosystem indicators are being researched and ecological and economic models are being developed. Additionally, among different possible ecosystem models for marine ecosystems, the Institute for European Environmental Policy indicated EwE as the most suitable for the development of scenarios to analyse future trends in marine biodiversity and changes in ecosystem services (Sukhedev, 2008).

Many Mediterranean ecosystem models were built in EwE (Coll and Libralato, 2012). Some cases of application of multi-species models other than EwE include the size-based model OSMOSE used for studies in the Gulf of Lion and Gulf of Gabes (Halouani et al., 2016b), the Model of Intermediate Complexity for Ecosystem assessments (MICE) for the main targets of the Adriatic Sea demersal fisheries (Angelini et al., 2016) and ATLANTIS in the Strait of Sicily (Peck et al., 2016). Moreover, the bio-economic models MEFISTO (Merino et al., 2007) and BEEMTOOL (Rossetto et al., 2015) have been developed. All these models have limited and restricted coverage and are specifically built for solving specific issues in the respective case studies. Modelling for fishery management purposes in the Mediterranean and in the world is still dominated by single-species models, especially in stock assessment evaluation (GFCM 2016b, c). Large-scale models in the Mediterranean are mainly the physical/oceanographic models (Somot et al., 2006; Oddo et al., 2009) and biogeochemical models (Petihakis et al., 2009; Lazzari et al., 2012; Macías et al., 2014; Teruzzi et al., 2014). The biogeochemical model describes the low trophic level processes with great detail, and this was further improved by coupling them online to the physical models (Cossarini et al., 2008). A significant step towards end-to-end ecosystem modelling in the Mediterranean was done by coupling online the high trophic level EwE-F model with the biogeochemical BFM model (Akoglu et al., 2014, Akoglu et al., 2015) and OSMOSE with Eco3M-NED in the Gulf of Gabes (Halouani et al., 2016b).

Applications of the ecosystem models not related to the Mediterranean include evaluations of EU's environmental assessments like Natura 2000 (Fretzer, 2016), Water Framework Directive (EU 2000/60) or the EU Marine Strategies Framework Directive (Piroddi et al., 2015a, b). Other, more specific, studies investigate the impacts of dumping sites on the ecosystem (Pezy et al., 2017) or ecosystem changes resulting from the installation of wind farms (Raoux et al., 2017). In fisheries management, the use of ecosystem models was proposed also to evaluate global maximal sustainable yields (Walters et al., 2005) as an improvement of single-species MSY estimates.

In the Mediterranean region, EwE models were used for various purposes (Coll and Libralato, 2012). Broader ecological questions included studies on degradation of sea food webs (Coll et al. 2008a), changes in ecosystem services (Liquete et al., 2016), effects of proliferation of non-indigenous species (Pranovi et al., 2003; Libralato et al., 2015; Corrales et al., 2017) and recovery of species populations and ecosystems (Fouzai et al., 2012; Coll et al., 2013a). Additionally, species trophic levels estimated through well-constructed ecosystem models were shown to be comparable to levels determined by the analysis of stable isotopes (Navarro et al., 2011; Albo Puigserver et al., 2015). Ecosystem models were built for specific areas or purposes, such as lagoons and enclosed basins (Palomares et al., 1993; Libralato et al., 2002; Pranovi et al., 2003; Libralato and Solidoro 2009; Piroddi et al., 2016), coral environment (Vassallo et al., 2016), deep sea environments (Moutopoulos et al., 2013; Tecchio et al., 2013; Tecchio et al., 2015), planktonic food-web (D'Alelio et al., 2016), or the assessment of impacts of aquaculture and fish farming (Solidoro et al., 2010; Piroddi et al., 2011; Forrestal et al., 2012; Bayle-Sempere et al., 2013; Izquierdo-Gomez et al., 2016). Fisheries interactions with marine protected areas (MPA, Libralato et al., 2006; Albouy et al., 2010; Colléter et al., 2015; Abdou et al., 2016; Prato et al., 2016) and the effects of Marine Protected Areas (Valls et al., 2012) were also studied. On the other hand, wide range area models, used to address fishery questions, were built for the Adriatic Sea (Zucchetta et al., 2003; Libralato et al., 2006; Coll et al., 2007; Coll et al., 2009; Barausse et al., 2009), Ionian Sea (Moutopoulos et al., 2013; Piroddi et al., 2016), South Catalan Sea (Coll et al., 2008c), Gulf of Gabes, Tunisia (Hattab et al., 2013) Aegean

Sea (Tsagarakis et al., 2010), Gulf of Lions (Bănaru et al., 2013) and the whole Mediterranean by combining several previously listed models (Piroddi et al., 2015). Applications of EwE have also included the study of the indirect effects of fishing (Pinnegar et al., 2004; Coll et al., 2006a, b; Moutopoulos et al., 2013) and the potential effects of gear modification (Coll et al., 2008b). Particular applications also involved applications of models to promote of regional ecosystem-based approach to fisheries (Coll et al., 2013) and attempts to create an end-to-end model (Libralato and Solidoro 2009; Akoglu et al., 2015). Spatially explicit models were developed to better address the different spatial-temporal distribution of resources and the fishing effort (Coll et al., 2016; Halouani et al., 2016a) and the development of MPAs (Zucchetta et al., 2003; Fouzai et al., 2012). However, attempts of operational ecosystem models used for fisheries management as in the North Sea (Mackinson and Daskalov, 2008) have not yet been done for the Mediterranean Sea.

1.4. Common Fisheries Policy landing obligation regulation in the Mediterranean

Discards represent unwanted fisheries catches of target and non-target marine species and are a management issue in fisheries worldwide (Kelleher, 2005; Tsagarakis et al., 2015; FAO, 2016). The EU recently included, in the reformed Common Fisheries Policy (CFP; EU 1380/2013; EU, 2013), measures that contrast the discarding practices, in particular the so called “landing obligation” (hereafter LO). According to this regulation and associated discard plans (EU, 2014; EU, 2016; EU, 2017), the unwanted catches of species that are subjected to catch limits (quotas) or minimum conservation reference sizes (MCRS) shall be retained on board of fishing vessels and landed (Table 2), their use for human consumption being forbidden.

For stocks regulated through the control of fisheries output, i.e. total allowable catches (TAC; like many stocks in the northern EU seas) discards are added to the marketable landings in the quotas. In these conditions, LO results in a strong incentive to improve selectivity in order to increase the marketable part as much as possible, and minimize the discards that have lower or no value (Condie et al., 2013; Vogel et al., 2017). The rationale is less clear for areas where management is mainly based on the input (effort) control rather than quotas, such as the EU Mediterranean Sea. In fisheries that are not regulated through quotas the LO represents an additional removal of biomass from the system. It is a management practice that neglects the ecological role of by-catch discarded to sea that contributes, to some extent, to fuel marine food webs through scavengers (Garcia Rivera et al., 2015). In systems where the scavenging activity is an important pattern for energy transfer in the food web (Pranovi et al., 2003; Moutopoulos et al., 2013), the landing of the previously discarded species can result in important food web changes (Heath et al., 2014; Fondo et al., 2015). Other than indirectly influencing the structure of populations at sea, therefore, the LO can influence the proportions of species in the landings of mixed fisheries and the overall economic value of catches. In such perspective the LO for species with minimum conservation reference size might have very critical effects (Sardà et al., 2015) such as i) the impoverishment of the marine ecosystem due to further energy removal (Garcia-Rivera et al., 2015), ii) an impairment of food web dynamics induced by effects on scavenger populations (Fondo et al., 2015; Kopp et al., 2016; Collie et al., 2016), iii) a further workload and infrastructure costs to fishermen (Batsleer et al., 2016; Villasante et al., 2016) and iv) an increase in the illegal market of undersized individuals (Bellido et al., 2017). Globally, therefore, LO risks being a no-win condition, with losses on both the ecological and the economical sides.

Table 2. Species regulated by the Annex III of the EC 1967/2006 regulation. Measures for fishes and mollusc bivalves are expressed as total length, other abbreviations: TL - total length, CL - carapace length.

Scientific name	Minimum size	Scientific name	Minimum size
Fishes			
Dicentrarchus labrax	25 cm	*Pagellus acarne*	17 cm
Diplodus annularis	12 cm	*Pagellus bogaraveo*	33 cm
Diplodus puntazzo	18 cm	*Pagellus erythtinus*	15 cm
Diplodus sargus	23 cm	*Pagrus pagrus*	18 cm
Diplodus vulgaris	18 cm	*Polyprion americanus*	45 cm
Engraulis encrasicolus	9 cm or 110 pieces/kg	*Sardina pilchardus*	11 cm or 55 pieces/kg
Epinephelus spp.	45 cm	*Scomber spp.*	18 cm
Lithognathus mormyrus	20 cm	*Solea vulgaris*	20 cm
Merluccius merluccius	20 cm	*Sparus aurata*	20 cm
Mullus spp.	11 cm	*Trachurus spp.*	15 cm
Crustaceans			
Homarus gammarus	300 mm TL, 105 mm CL	*Palinuridae*	90 mm CL
Nephrops norvegicus	20 mm CL, 70 mm TL	*Parapenaeus longirostris*	20 mm CL
Mollusc bivalves			
Pecten jacobaeus	10 cm	*Venus spp.*	25 mm
Venerupis spp.	25 mm		

For species regulated through catch control, i.e. total allowable catches (TAC, like many species in the northern EU seas), discards sum to the marketable landings in the quotas. In these conditions, LO results in a strong incentive to improve selectivity to increase the marketable part of the imposed quotas as much as possible. However, the vast majority of exploited Mediterranean species are not regulated by catch limits but by effort control, gear specifications and exclusion zones (Longo et al., 2015). In this situation the landing obligation for species with minimum sizes may have very critical effects as outlined above and in Sardà et al. (2013).

In principle, the LO represented a tool at the disposal of the EU to promote the improvement of gear selectivity even in areas where fisheries regulations are not based on TACs and quota systems. This may be partially true, but since discardless catches are unrealistic, the LO will necessarily represent an additional uptake of biomass from the system for species that, in many cases, already suffer for excessive exploitation (Garcia-Rivera et al., 2015). Moreover, in systems where scavenging activity is an important pathway for energy transfer in the food web, landing of previously discarded species can result in important food web changes resulting in altered population structures. This indirectly influences catches and potentially the overall economic value of a fishery by changing the proportions of species in its catch portfolio. The overall effect can be positive in cases where species increasing after LO are those highly priced by the market. In addition, one needs to consider also the large extra workload and costs that fishermen need to sustain in order to follow LO requirements (Villasante et al., 2016). The fact that discarded species may have a market other than direct human consumption might result in benefits that can compensate extra costs, but this needs to be evaluated quantitatively. Although these aspects are reported in the recent literature (Tsagarakis et al., 2013), they are seldom considered in a quantitative ecosystem approach framework (sensu Pikitch et al. 2004; Cochrane and Young 2008).

Holistic approaches such as ecosystem models allow to integrate the representation of complex food web dynamics (e.g. Piroddi et al., 2016), the exploitation by different gears and their specific discards and by catch (e.g. Moutopoulos et al., 2014) and to include information on prices and costs (Sumaila et al., 2000). These approaches permit to analyse the direct and indirect effects of the landing obligation in the ecosystem allowing quantifying both ecological and economic advantages and disadvantages of medium and long term policies (see for example Walters and Martell, 2003).

1.5. Aims

The present thesis has the objective of developing an ecosystem model of the North-East Adriatic Sea (NEAS), as exemplificative of a Mediterranean system exploited by mixed fisheries targeting a large set of species, and test for the potential of the tool to be used as an operational tool for fisheries management in the area. The developed model is thus tested through i) an in depth analysis of its precision and accuracy in representing field data, ii) an application to quantify ecosystem and socio-economic effects of a new management regulation (the landing obligation), iii) a scenario analysis of adaptation measures that might be applied to optimize fishing strategies.

Therefore, Chapter 2 will give a brief overview of the data collected for the NEAS model while the model suite Ecopath with Ecosim will be described in Chapter 3. The integration of the data into a mass-balanced Ecopath model and its preliminary analysis is provided in Chapter 4. On the other hand, the integration of the observed time series into a time-dynamic Ecosim model and the common dynamic parameterisation will be illustrated in Chapter 5. Additionally, in the same chapter a preliminary analysis of the time-dynamic model sensitivity will be carried by two tests. The first will aim to investigate the influence of variation of basic mass-balanced parameters and the initial conditions on the estimates of functional groups' biomass by using a Monte Carlo analysis. The second test aims to investigate the influences of different fishing effort regimes on the different functional groups. In Chapter 6 an ensemble of models will be developed by using different fitting strategies with the goal of representing the intrinsic variability of food web responses. The ensemble's capabilities for further operational uses will be validated by estimating different measures of the goodness of fit, such as bias, precision, correlation and squared deviations. The effects the Landing Obligation may have on Mediterranean ecosystem and fisheries socioeconomics will be evaluated in Chapter 7 through a NEAS case study using an operational ensemble of models. In Chapter 8 alternative management solutions aiming at the same goals as the Landing Obligation will be evaluated through different fisheries adaptation scenarios. In the same chapter a multispecies management scenario will be evaluated with the NEAS model. Final discussion and conclusion will follow in Chapter 9.

2. The North-East Adriatic Sea (NEAS)

2.1. Study area

At the northernmost part of the central Mediterranean, the Northern Adriatic has many peculiarities that make it a diversified ecosystem with a rich historical, ecological and social background. The semi-enclosed basin has a counter-clockwise circulation influenced by the river outflows along the western coast, primarily the Po River (Zavatarelli et al., 1998; Malačič and Petelin, 2009). The contribution of the nutrient loads from the rivers and remineralisation processes are responsible for the eutrophic waters and high primary production along the west and the northern coast (Giani et al., 2012). On the contrary, the eastern waters have oligotrophic and mesotrophic characteristic, where the limited riverine inputs limit the availability of nutrients for primary production, therefore creating an east-west gradient. The input of nutrients in the Adriatic Sea is mostly due to anthropogenic activities, strongly influencing eutrophication and primary productivity, and is of the same order as the regenerated amounts during the seasonal cycle. High inputs of organic matter from the rivers also sustain the high macro-fauna biomasses. The Adriatic Sea has one of the widest shelves in the Mediterranean: its muddy and sandy bottoms with few rocky outcrops are suitable for trawling activities and have been historically exploited (Botter et al., 2006; Fortibuoni et al., 2010). In the last decade, the management of Adriatic shared stocks has been improved through joint assessments of shared resources (e.g. small pelagics and hake) (GFCM, 2016b; GFCM, 2016c), contributing to the establishment of the needed shared fishery policies between the Adriatic countries (Vrogč et al., 2004).

In the last century, the basin has gone through substantial changes in response to several triggering events. Increasing eutrophication was observed from the 1930s, and a series of anoxic events occurred during the 1960s, 1970s and 1980s, some of which (e.g. 1977 and 1989) also interested the entire Adriatic basin (Giani et al., 2012). In the Gulf of Trieste, after reaching peak biomasses in mid 1970s, a mass mortality in 1983 caused a decline of the epi-benthic communities that never recovered because of an anoxic crisis in the 1988 and the trawling activity (Giovanardi et al., 1998; Pranovi et al., 2000). To reduce the eutrophication of coastal waters phosphorous has been banned as a water softener in soaps by the Italian law from the mid-1980s. Together with the improvement of sewage treatment and the reduction of river outflows (Cozzi et al., 2012), this resulted in a trend of partially cultural oligotrophication (Mozetič et al., 2010). Together with the warming of surface Adriatic waters (Russo et al., 2009), it also caused a reduction of chlorophyll-a in the basin, especially in the western river-influenced waters, and shifts of the phytoplankton and zooplankton community compositions (Mozetič et al., 2010). Many demersal resources had already been exploited since the second world war, especially sharks and rays (Jukić-Peladić et al., 2001; Feretti et al., 2013) and the fishing fleet was at its maximum in the mid-1980s (Piroddi et al., 2015). Owing to the reduced primary production and the collapse of the fishery resources (Fortibuoni et al., 2017), the fishing fleets suffered a major crisis (Piroddi et al., 2015). For example, reduced riverine nutrients and primary production together with the overfishing may have influenced the decline of the clam fisheries (Morello et al., 2005; Romanelli et al., 2009). The spreading of non-indigenous species caused by shipping or aquaculture activities adds to the present pressures and may interfere with existing communities and the balance of the food webs (Occhipinti-Ambrogi, 2002; Pranovi et al., 2003; Orlando Bonaca, 2010).

The Northern Adriatic is, thus, a suitable case study to approach Mediterranean complexities systematically. Moreover, the recent EU fishery regulation promotes the regionalisation of fishery management and the bottom-up approach so to account for local specific characteristics. In this way local entities may have an important role in the future management and decision making processes and the availability of an operational model might represent a possible support for decision makers.

The study site covers an area of approximately 1000 km^2 north of the imaginary line connecting the Punta Tagliamento (Italy) and Cape Savudrija (Croatia). Although the Grado Marano lagoon has a significant ecological role and is connected to the case study, it was excluded from the present case study because of insufficient data (Figure 2). The study area presents a remarkable biological and geological variability. It has a highly diversified coast, a sandy coast at the west until Monfalcone, followed by a rocky coast until Trieste and boulders and flysch stones along the Slovenian coast (Lipej et al., 2003). This area is characterized by shallow depths (maximum 25 m), sandy bottom (Harris, 2008), and highly productive waters (Fonda Umani et al., 2012). Water quality in NEAS is strongly influenced by riverine input (Solidoro et al., 2007, Solidoro et al., 2009) that, in conjunction with local winds, drives the water circulation (Querin et al., 2006). River inputs also sustain the high productivity of the area (Cossarini and Solidoro, 2008). The NEAS hosts high biodiversity habitats like rocky outcrops, locally called “trezze”, (Gordini, 2009) and seagrass meadows (Falace et al., 2015), that represent important spawning and nursery areas.

Several protected areas are located in the study area: Strunjan Nature Reserve, Debeli rtič and Cape Madona Nature Monuments in Slovenia (Lipej et al., 2003) and Miramare Marine Protected Area (MPA), Natura 2000’s Communitarian Interest Sites (“trezze di San Pietro e Bardelli” and

“Cavanata di Monfalcone”), Natura 2000’s Special Conservation Site “Valle Cavanata e Banco Mula di Muggia” and the Isonzo estuary Natural Reserve in the Friuli Venezia-Giulia region.

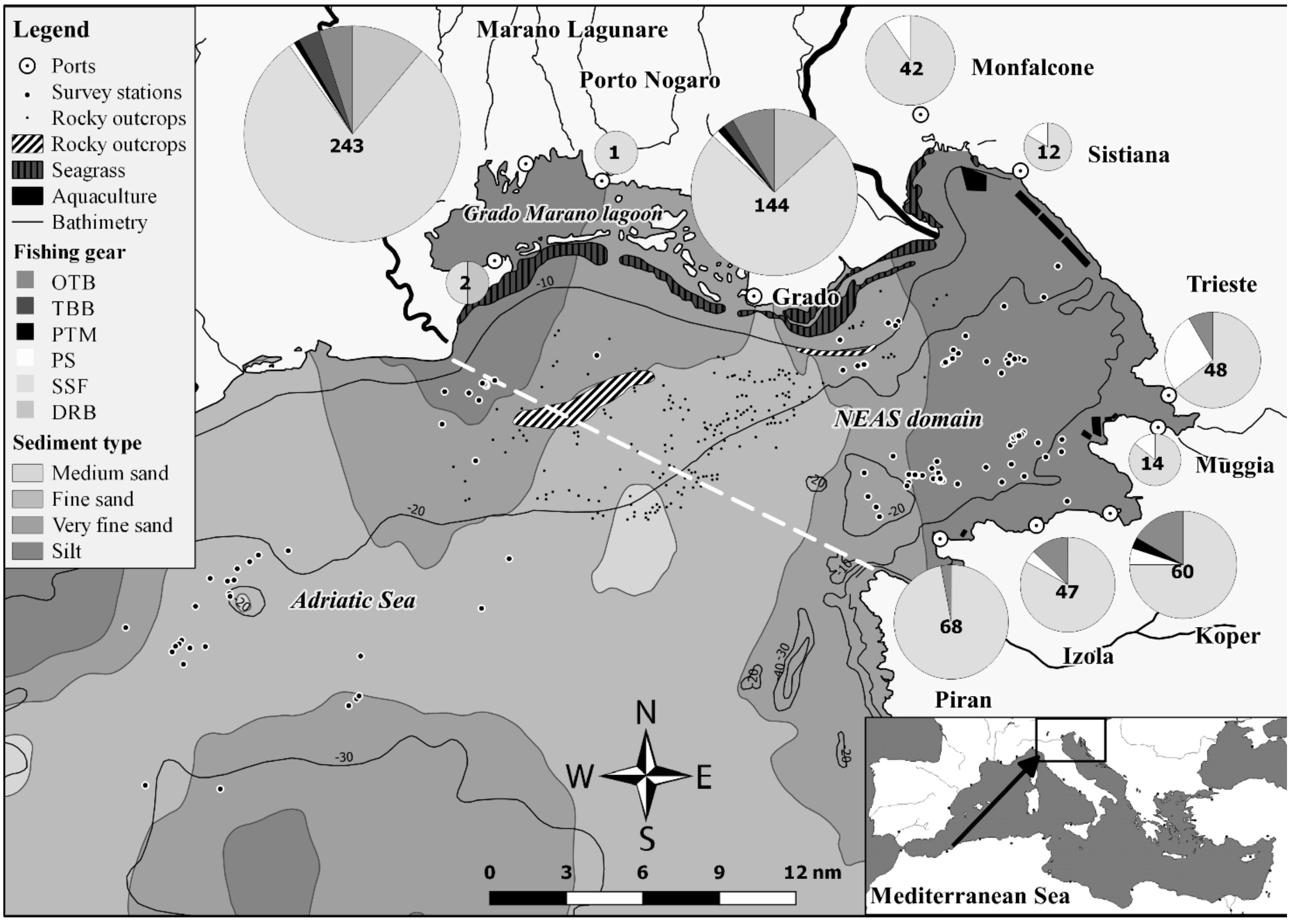

Figure 2. Main features of the North-East Adriatic Sea (NEAS), including sediment and bathymetry. The area within pie charts is proportional to the number of fishing vessels by gear in each port, total number of fishing vessel is reported in the pie. Fishing gear codes: bottom otter trawl (OTB); “rapido” beam trawl (TBB); midwater pelagic trawl (PTM); “lampara” purse seine (PS); small-scale fisheries (SSF); hydraulic dredges (DRB).

2.2. Ecosystem of the North-East Adriatic Sea

The NEAS area is characterised by a high biodiversity and hosts a complex ecosystem with various types of habitats: sandy bottoms, marine phanerogams and inshore various microhabitats. The area is mainly characterized by a sandy bottom habitat, covering the largest areas from the deeper parts and reaching the shallow coastline. The macro-epifauna of sandy bottoms is dominated by ophiurids (*Ophiotrix fragilis* and *Ophiura ophiura*), tunicates (*Phallusia mammilata*, *Microcosmus spp*., *Molgula spp*. and *Pyura spp*.), sponges (*Suberites domuncula* and other species), gastropods (*Aporrhais pespelicani*, *Hexaplex trunculus* and *Bolinus brandaris*), sea urchins (*Psammechinus microtuberculatus*, *Paracentrotus lividus*, *Sphaerechinus granularis*), sea stars (*Astropecten irregularis*), holothurians (*Cucumaria planci* and *Holothuria spp*.) and shellfish (*Aequipecten opercularis* and *Chlamys spp*.) (Celić, 2008; Santelli et al., 2017). Marine phanerogams also represent an important habitat in the study area (EMODnet, www.emodnet-seabedhabitats.eu). Extensive

Cymodocea nodosa and *Zoostera marina* meadows are located along the western coast and other smaller meadows can be found along the Slovenian and the northern rocky coasts. The once flourishing *Posidonia oceanica* now has the smallest coverage among the marine phanerogams and is restricted to three isolated meadows: two in front of Grado's navigation channel (Falace et al., 2015) and one along the coastline between Izola and Koper (Orlando Bonaca et al., 2004). Other inshore microhabitats are the associations with *Cystoseira spp.*, coralline algae *Halophytis incurvus* and *Corallina granifera* settlements, *Wagnelia penicilata* and *Padina pavonica* settlements and boulders covered with algal turf.

The study area is an important spawning and nursery area for several marine species (e.g., the cuttlefish *Sepia officinalis* and the squid *Loligo vulgaris*). The sandy bottoms are inhabited also by small-sized fish species (Gobiidae, Blenniidae, Callionymidae and Pleuronectiformes). Coastal planktivorous species like the smelt *Atherina spp*. and *Chromis chromis* can be found along the coastline and associated to the marine phanerogams meadows (Orlando Bonaca and Lipej, 2004). Other frequent dwellers of the coastal microhabitats belong to fish families like the Blenniidae, Centracanthidae, Gobiidae, Labridae, Moronidae, Mugilidae, Sciaenidae, Serranidae, Sparidae and Syngnathidae (Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca and Lipej, 2005). The rocky outcrops "trezze" present in the area have an important role as habitats or refuge for hard bottom fish and invertebrate communities (Borme et al., 2011).

Other important residents and predators on fish and cephalopods are the bottlenose dolphin *Tursiops truncatus* (Genov et al., 2008) and the marine birds nesting in the area, primarily the great cormorant *Phalocrocorax carbo,* the yellow-legged seagull *Larus michahelis* and *Chroicocephalus ribidibundus* (Zenatello et al., 2014).

2.3. Primary production and plankton changes

The trend of primary production in the NEAS area was obtained from satellite observations data of surface chlorophyll-a (available at marine.copernicus.eu, Volpe et al., 2007; Santoleri et al., 2008). The interaction with macroalgal pigments in shallow waters produced extremely high chlorophyll-a values, therefore, the nearest sampling points to the shore were excluded using a spatial filter. The monthly trend of chlorophyll-a in the NEAS area was estimated using the median value (Figure 3). Seasonality between diatoms and dinoflagellate were determined by decomposing the monthly trend into a diatom autumn/winter peak and a dinoflagellate spring/summer peak (Cossarini and Solidoro, 2008).

The primary productivity rates were obtained by correcting the monthly cycle trends in Cossarini and Solidoro (2008) in order to fit the phytoplankton biomass trends (Figure 3). The chlorophyll-a shows lower concentration in the period 2004-2009 and higher concentrations from 2010. The peaks of chlorophyll-a in 2010 and 2014 match the observed increases in diatom and dinoflagellate densities from 2009 in the Gulf of Trieste (Lipitzer et al., 2015).

Trends for Macroalgae and seagrass coverage and productivity were not available, so a repeated monthly cycle as in Peduzzi and Vukovič (1990) was used with a constant mean yearly production rate.

Bacterioplankton consumption rates on particulated organic matter (POM) and bottom detritus were taken from Cossarini and Solidoro (2008).

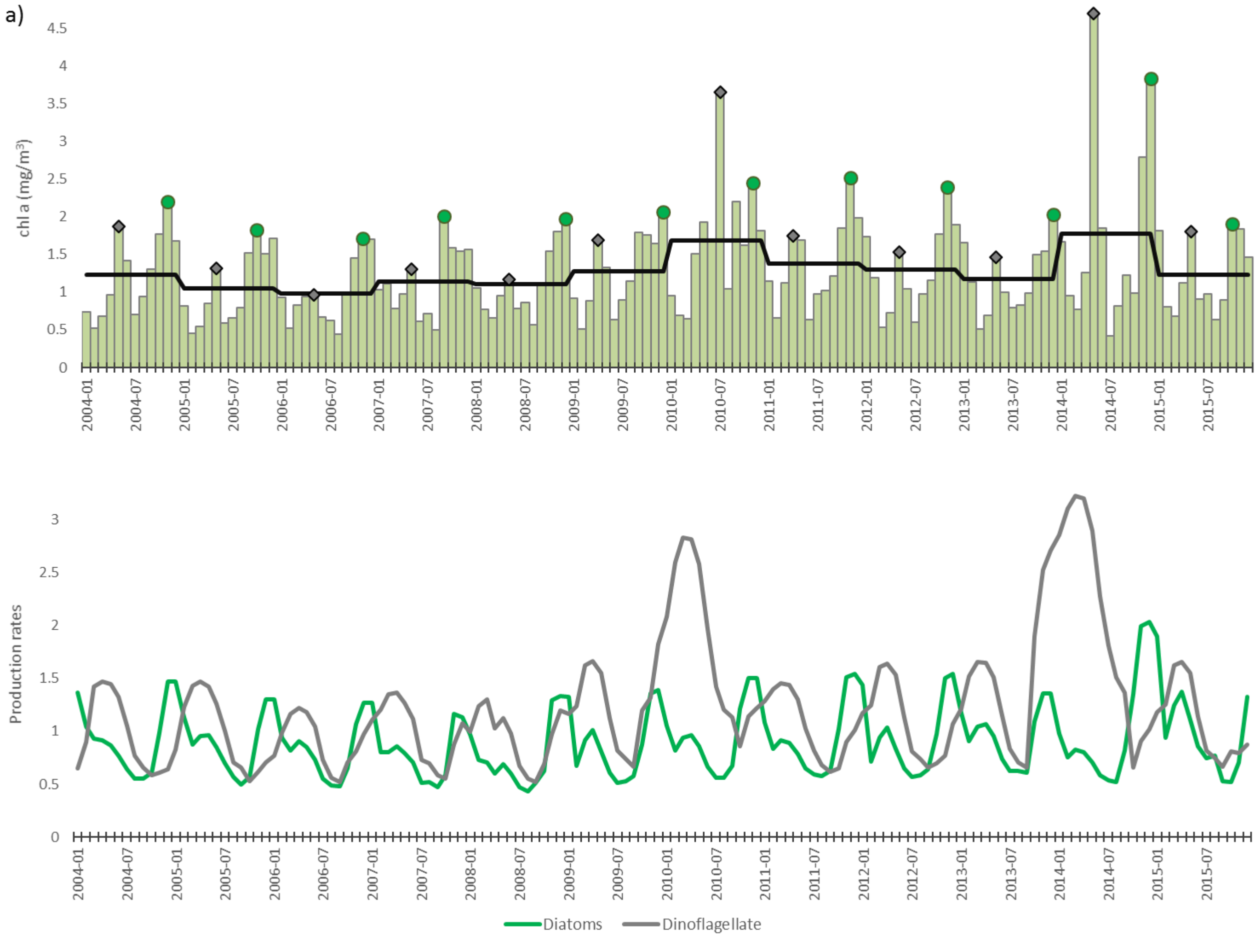

Figure 3. Monthly trend of the chlorophyll-a and the diatom and dinoflagellate relative primary production rates in the NEAS area. a) Chlorophyll-a monthly concentration is in light green bars and the yearly mean value is the black line. The seasonal peaks were decomposed for diatoms in green dots and dinoflagellates in grey diamonds. b) Diatoms and dinoflagellate monthly relative primary production rates, first year mean production rate was 479.5 t km^{-2} y^{-1} for diatoms and 162.3 t km^{-2} y^{-1} for dinoflagellates.

2.4. Biomass of ecosystem components

Time series (2005-2014) of biomass for Marine birds were obtained from seabird census (Zenatello et al., 2014) considering an average individual size (*Phalocrocorax carbo*: 3 kg; *Casmerodius albus*: 1 kg; *Ardea cinerea*: 1.5 kg; *Chroicocephalus ridibundus*: 0.5 kg; *Larus melanocephalus*: 0.3 kg; *Larus canus*: 0.4 kg; *Larus argentatus*: 1.5 kg). Mesozooplankton biomass was reconstructed using information from Mozetič et al. (2012) and microzooplankton from Lipitzer et al. (2015). Biomasses for fish and other benthic invertebrates were estimated for the period 2005-2014 from the common sole rapido trawl survey (SoleMON, 34 sampling station; Grati et al., 2013) and the Mediterranean international trawl survey (MEDITS, 78 sampling stations; Bertrand et al., 2002). Data from the trawl surveys were divided in three subareas (southern part of the domain; north-eastern part of the domain; central western part) on the basis of data coverage and seabed characteristics that highly influence density and composition. The MEDITS was used to estimate the biomass of sharks, rays, demersal fish and demersal invertebrates since the use of the bottom otter-board trawl has good sampling capabilities for these species. On the other hand, the use of a rapido trawl in the SoleMON survey made it suitable to assess the biomass of flatfish and benthic macro-epifauna. Relative density by species obtained from trawl surveys were transformed in absolute biomass using catchability parameters. The catchabilities were determined by Lopez (2013) for several species though a comparison between MEDITS trawl survey average densities and stock assessment results for the Thyrrhenian sea. Moreover, for benthic and some demersal species, catchabilities were determined by comparing MEDITS average densities with catches from scientific trawl surveys conducted with beam trawl gears in the same areas (Lopez, 2013). These catchabilites were compared values reported in the literature (Alemany and Álvarez, 2003; Mahévas et al., 2011; Fiorentino et al., 2013).

Specie were aggregated in functional groups that would be later used for building the ecosystem model in Chapter 4 (Table 3).

Table 3. List of functional groups with reference time series and the species composing them.

MarBird	Marine birds *Ardea cinerea, Casmerodius albus, Chroicocephalus ridibundus, Larus argentatus, Larus canus, Larus melanocephalus, Phalocrocorax carbo*
Elasm	Elasmobranchii *Dasyatis pastinaca, Mustelus mustelus, Mustelus punctulatus, Myliobatis aquila, Pteroplatytrygon violacea, Raja asterias, Raja clavata, Scyliorhinus stellaris, Squalus acanthias, Torpedo torpedo*
SmallPel	Small pelagic fish *Engraulis encrasicolus, Sardina pilchardus, Sprattus sprattus*
BenthPel	Benthopelagic fish *Boops boops, Merlangius merlangus, Scomber japonicus, Scomber scombrus, Trachurus mediterraneus, Trachurus trachurus, Trisopterus capelanus*
CoastPla	Coastal planktivorous fish *Alosa fallax, Aphia minuta, Atherina spp., Chromis chromis*
PelPisc	Pelagic piscivorous fish *Auxis rochei, Dicentrarchus labrax, Euthynnus alletteratus, Lichia amia*
InvFeed	Invertebrate feeding fish *Diplodus annularis, Diplodus sargus sargus, Diplodus vulgaris, Spondyliosoma cantharus, Sparus aurata*
Detritiv	Detritivorous fish Mugilidae
Herbiv	Herbivorous fish *Sarpa salpa*
Flatfish	Flatfish *Arnoglossus laterna, Arnoglossus thori, Citharus linguatula, Platichthys flesus, Buglossidium luteum, Monochirus hispidus, Pegusa lascaris, Solea aegyptiaca, Solea solea, Pegusa impar, Synapturichthys kleinii, Microchirus variegatus*
Benthiv	Benthivorous fish Blenniidae, Callionymidae, Cepolidae, Gobiidae, Syngnathidae, Triglidae, *Conger conger, Lithognathus mormyrus, Mullus barbatus, Mullus surmuletus, Pagellus acarne, Pagellus bogaraveo, Pagellus erythrinus, Scorpaena notate, Serranus hepatus, Spicara spp., Umbrina cirrosa*
DemPisc	Demersal piscivorous fish *Merluccius merluccius, Trachinus draco, Uranoscopus scaber, Scophthalmus maximus, Scophthalmus rhombus, Scorpaena porcus, Scorpaena scrofa, Zeus faber*
Cephal	Cephalopoda *Alloteuthis media, Eledone moschata, Loligo vulgaris, Sepia elegans, Sepia officinalis, Sepiola spp.*
Bivalv	Bivalvia *Acanthocardia tuberculate, Aequipecten opercularis, Atrina fragilis, Callista chione, Corbula gibba, Flexopecten glaber, Pecten jacobaeus*
AnnWorm	Annelida & Other Worms *Aphrodita aculeate, Aspidosiphon (Aspidosiphon) muelleri muelleri* and other Errantia and similar phyla
DecaSto	Decapoda & Stomatopoda Anomura, Astacidea, Brachyura, Caridea, Penaeoidea, *Squilla mantis*, and other Decapoda
Gastrop	Gastropoda *Aporrhais pespelecani, Bolinus brandaris, Hexaplex trunculus, Turritella communis* and other Gastropoda
Echinod	Echinodermata *Astropecten irregularis, Holothuria (Holothuria) tubulosa, Leptopentacta spp., Ocnus planci, Ophiothrix fragilis, Ophiura ophiura, Paracentrotus lividus, Psammechinus microtuberculatus, Sphaerechinus granularis* and other Echinodermata
OthBenth	Other benthic filter feeders *Botryllus schlosseri, Calliactis parasitica, Microcosmus vulgaris, Molgula spp., Phallusia mammillata, Polycarpa spp., Pyura spp., Suberites domuncula, Tethya aurantium*, Cnidaria, Porifera, Tunicata and other sessile fauna
MesoZoo	Mesozooplankton
MicroZoo	Microzooplantkon
Phy1Dino	Phytoplankton 1 – Dinoflagellate
Phy2Diat	Phytoplankton 2 – Diatoms

Each species' average biomass was obtained as the weighted biomass for the area of the sampled subunits. In addition, the biomass variation was estimated for SoleMON and MEDITS data when two or more subareas were sampled that year. The yearly total variation was calculated as a weighted sum of each area's variation as described in the following equations (Eq.: 1-8):

$$\bar{\sigma} = \sqrt{\frac{\sum_{i=1}^{N}\left(w_i \cdot \sigma_i^2\right) + \sum_{i=1}^{N}\left[w_i \cdot \left(mean_i - \overline{mean}\right)^2\right]}{\sum_{i=1}^{N} w_i}} \tag*{Eq. 1}$$

$${\sigma_{subarea}}_{sp,a,y,fg}^{2} = \frac{\sum_{st=1}^{n}\left(B_{sp,st,a,y,fg} - \overline{{B_{subarea}}_{sp,a,y,fg}}\right)^2}{{N_{stations}}_{a,y}} \tag*{Eq. 2}$$

where:

$$\overline{{B_{subarea}}_{sp,a,y,fg}} = \frac{\sum_{st=1}^{n} B_{sp,st,a,y,fg}}{{N_{stations}}_{a,y}} \tag*{Eq. 3}$$

$$\overline{w.{\sigma_{subarea}}_{sp,y,fg}^{2}} = \frac{\sum_{a=1}^{n}\left({\sigma_{subarea}}_{sp,a,y,fg}^{2} \cdot S_{a,y}\right)}{\sum_{a=1}^{n} S_{a,y}} \tag*{Eq. 4}$$

$$w.{\sigma_{means.subarea}}_{sp,y,fg}^{2} = \frac{\sum_{a=1}^{n}\left(\left(\overline{{B_{subarea}}_{sp,a,y,fg}} - \overline{{B_{domain}}_{sp,y,fg}}\right)^2 \cdot S_{a,y}\right)}{\sum_{a=1}^{n} S_{a,y}} \tag*{Eq. 5}$$

where:

$$\overline{{B_{domain}}_{sp,y,fg}} = \frac{\sum_{a=1}^{n}\left(\overline{{B_{subarea}}_{sp,a,y,fg}} \cdot S_{a,y}\right)}{\sum_{a=1}^{n} S_{a,y}} \tag*{Eq. 6}$$

$${\sigma_{domain}}_{sp,y,fg}^{2} = \overline{w.{\sigma_{subarea}}_{sp,y,fg}^{2}} + w.{\sigma_{means.subarea}}_{sp,y,fg}^{2} \tag*{Eq. 7}$$

$${\sigma_{domain}}_{y,fg} = \sqrt{\frac{\sum_{sp=1}^{n}\left({\sigma_{domain}}_{sp,y,fg}^{2} \cdot \overline{{B_{domain}}_{sp,y,fg}}\right)}{\sum_{sp=1}^{n} \overline{{B_{domain}}_{sp,y,fg}}}} \tag*{Eq. 8}$$

Where $B_{sp,st,a,y,fg}$ is the sampling station's biomass, $\overline{{B_{subarea}}_{sp,a,y,fg}}$ the subarea's mean biomass and $\overline{{B_{domain}}_{sp,y,fg}}$ the model domain's mean biomass. The specie, sampling station, area, year and the functional group are indicated respectively as *sp*, *st*, *a*, *y* and *fg*. The resulting species' average variation in the modelled domain ${\sigma_{domain}}_{sp,y,fg}^{2}$ equals to the surface weighted sum of subarea's variations ($\overline{w.{\sigma_{subarea}}_{sp,y,fg}^{2}}$) and the variation of subareas' means ($w.{\sigma_{means.subarea}}_{sp,y,fg}^{2}$). The ${\sigma_{domain}}_{y,fg}$ is the resulting functional groups' standard deviation weighted for each species' biomass contribution. MEDITS biomasses of small pelagic species were deemed unreliable since the sampling gear is unspecific; therefore, no reference biomass data was estimated.

Outliers were excluded from the reference biomass time series and the results are shown in Figure 4. A decreasing trend for biomasses in the NEAS domain can be observed for Marine birds, Benthivorous fish, Cephalopoda, Decapoda & Stomatopoda, Gastropoda, while an increasing trend resulted for Elasmobranchii, Flatfish, Echinodermata, Other benthic filter feeders and Microzooplankton.

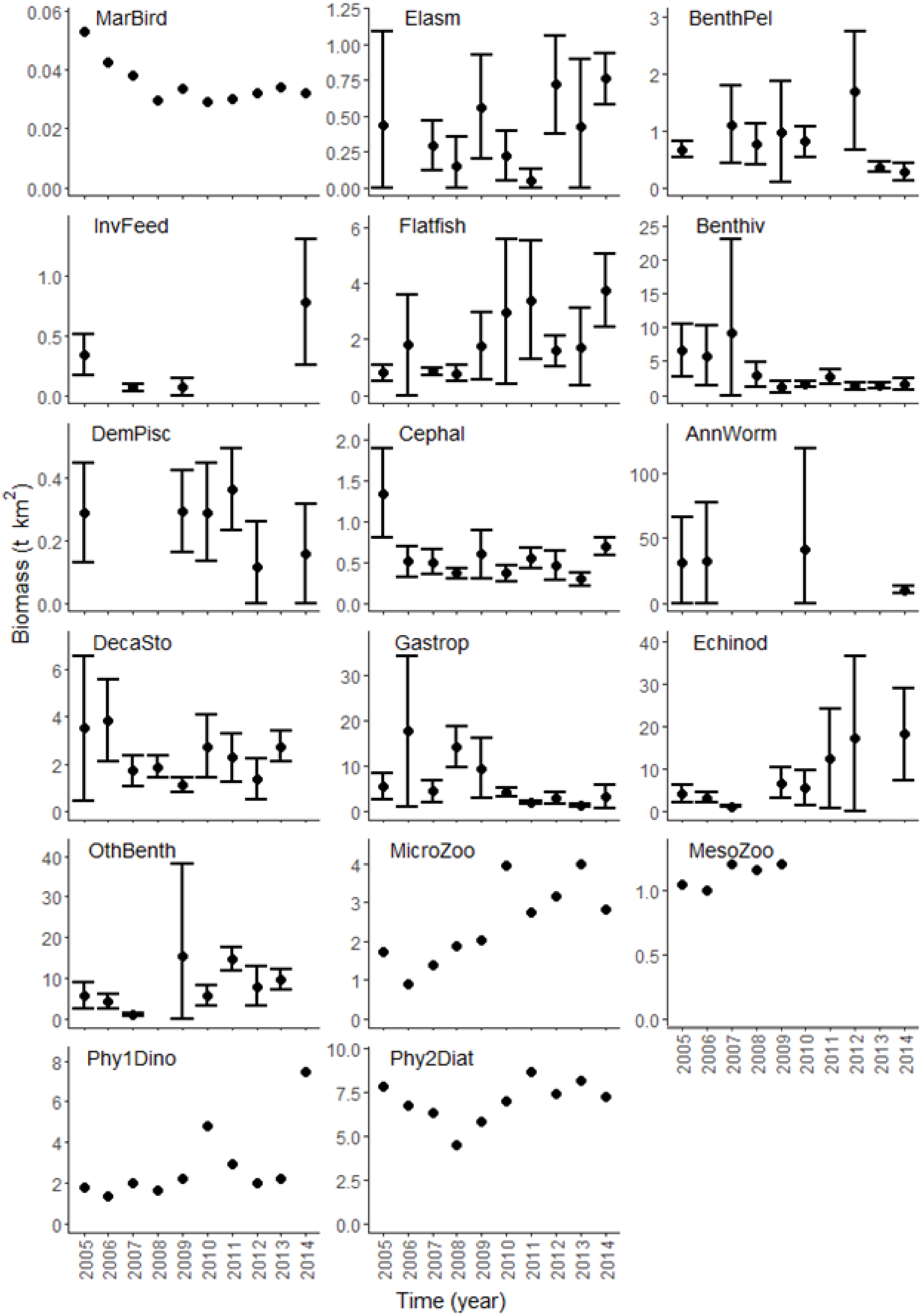

Figure 4. Biomass data and the standard deviation for the NEAS domain. Functional group names are reported in Table 3.

2.5. Fisheries in the North-East Adriatic Sea

The NEAS provides several goods and services including marine resources exploited by mixed fisheries and extensive mariculture (Solidoro et al., 2010). Fishing is an important activity in the study area since centuries (Faber, 1883). It includes low impact small-scale fisheries (SSF; set nets, traps, lines and hooks), as well as bottom otter trawling and “rapido” trawling (OTB and TBB, respectively; mainly in Grado, Marano Lagunare, Koper and Izola). Moreover, hydraulic dredges (DRB) for harvesting clams (*Callista chione*, *Chamelea gallina*) operate in the Italian coastal areas (Biondi and Del Piero, 2012). Small pelagics are exploited by purse seines (PS) using lights for attracting fish (“lampara” in Trieste, Muggia, Izola and Koper) and midwater pelagic trawl (PTM) (“volante” in Marano Lagunare, Grado and until 2012 in Izola) (Morello and Arneri, 2009). Overall, although “lampara” purse seine can be quite selective, these fisheries are generally multi-target and, especially bottom trawlers and hydraulic dredges, can have considerable quantities of discards, also composed by small-sized species or juveniles (Pranovi et al., 2001; Morello et al., 2005; Botter et al., 2006; Raicevich, 2008)

According to the number of fishing vessels from harbour census and EU Fleet Register data (Fleet Register, http://ec.europa.eu/fisheries/fleet/index.cfm) there were 681 fishing vessels active in 2005. Gear assignment by EU Fleet Register was reviewed by local harbour census data. More than half of the fishing vessels are based in Marano Lagunare (243 vessels) and Grado (144 vessels), while the rest is distributed in Piran (68 vessels), Koper (60 vessels), Trieste (48 vessels), Izola (47 vessels) and Monfalcone (42 vessels), and a minority in Muggia (14 vessels), Sistiana (12 vessels), Lignano (2 vessels) and Porto Nogaro (1 vessels) (Figure 2). The small scale fishery is the dominant fishing activity in the area (79.4%) and in all fishing ports. The two second most frequent fishing gears are the dredges fishing in the study area (47 vessels) based in Grado, Marano Lagunare, and the bottom otter-board trawlers (46 vessels), mainly in ports of Grado, Marano Lagunare, Koper and Izola. The purse seines (“lampara”) are 30, located primarily in Trieste and distributed throughout the bay. Rapido trawlers and midwater pair pelagic trawlers are the smallest fleets, respectively 11 and 6 vessels, both located in Grado and Marano Lagunare, and an additional pair of midwater pair pelagic trawlers in Izola.

2.6. Fisheries catches in the NEAS

Reconstruction of the catch reference data was based on landings data in IREPA reports, Trieste’s fish market (Fortibuoni et al., 2010) and Slovenia’s reconstructed landings (http://www.biosweb.org/?task=stat, Bolje et al., 2014); discards were determined proportionally to the landings using information available in the literature (Pranovi et al., 2001; Morello et al., 2005; Botter et al., 2006; Raicevich, 2008). Reference data are available in Figure 5. Decreasing catches in the NEAS domain were resulting for Elasmobranchii, Small pelagic fish, Benthivorous fish, Bivalves, Decapoda & Stomatopoda and Gastropoda, while an increasing trend was observed for Herbivorous fish and Demersal piscivorous fish. It is worth noting that catches for Annelida & other worms, Echinodermata and Other benthic filter feeders are composed only by discards and their changes was estimated on the basis of the total landings using literature information (Pranovi et al., 2001; Morello et al., 2005; Botter et al., 2006; Raicevich, 2008).

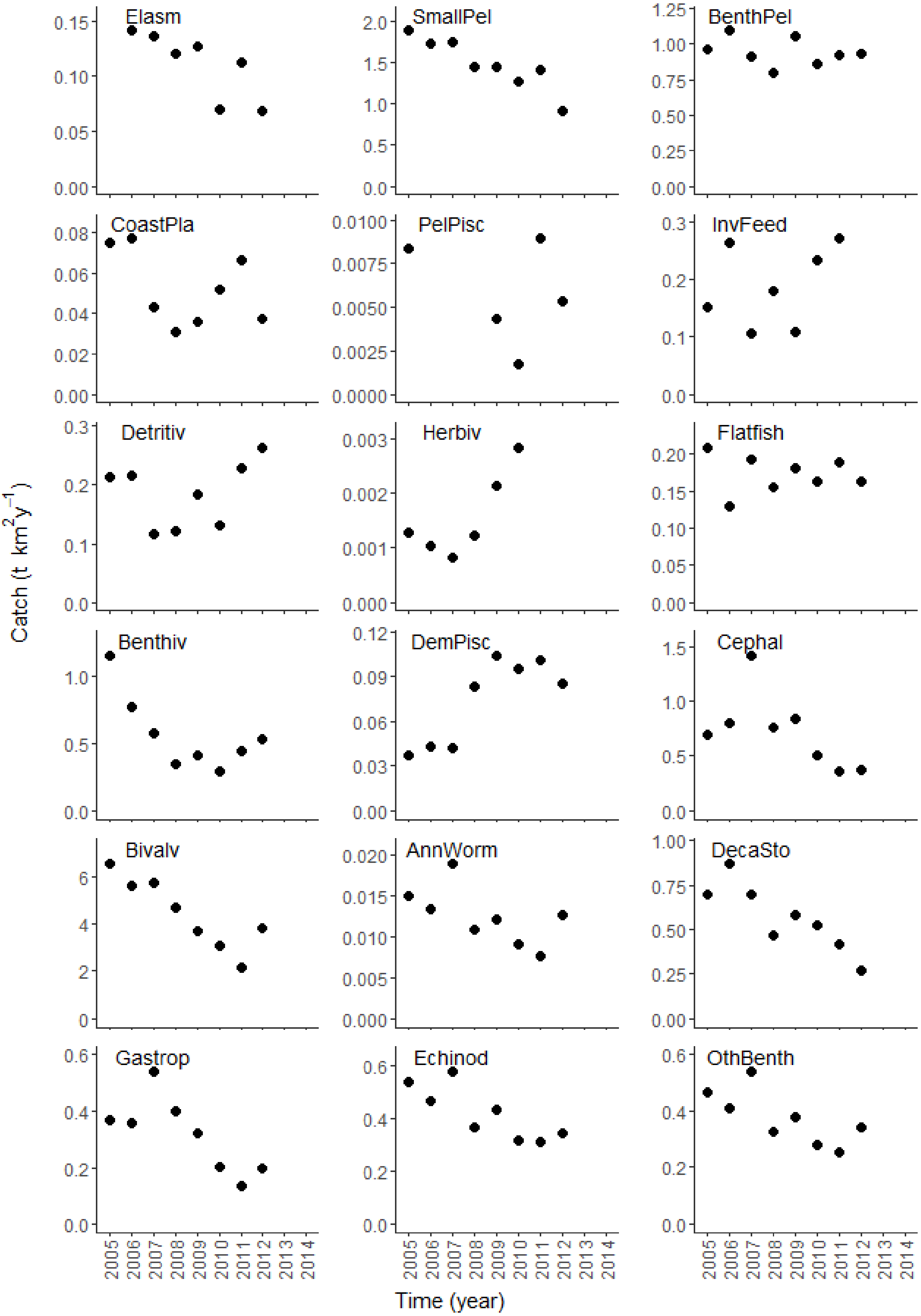

Figure 5. Catch data for the NEAS ecosystem grouped by main functional groups. Catches include commercial landings and dead discards. Functional group names are reported in Table 3.

2.7. Analysis of fishing effort indicators

Fishing effort for Italy's and Slovenia's fleets was calculated on the basis of the EU Fleet Register data on fleet characteristics (2005-2015) and the number of days at sea for each fleet as declared in IREPA reports (2005-2012; the last three years, 2013-2015, were estimated from the its previous three years' average). Thus, days at sea with fleet characteristics over time were used to define changes in fishing effort. Fleet characteristics included the number of vessels, LOA (Length Over All, in meters) and engine power (kW). Although all measures of effort were considered (see Chapter 5) the reference effort was considered by fleet according to relevant characteristics of the metier.

For OTB, TBB and PTM (Eq. 9), the cubic LOA (Length Over All, in meters) was considered more reliable and used as a descriptor of fishing capacity (Hsu, 2003) since the engine power (kW) is often non correctly reported and the gross tonnage (GT) expresses the official useful operating volume according to vessel's blueprints and not the real usage. Number of vessels was considered a good indicator of fishing capacity for PS, SSF and DRB, fishing technique and fishing device being independent from vessels size or, for some fishing activities a same daily quota being present for all fishing vessels (Eq. 10). The determination of fishing effort (*E*) by year (*y*) and fleet (*fl*) was based on individual vessel's (*v*) registered service activity and specifications (LOA and gear; EU Fleet Register), and fleet fishing yearly activity derived from monitoring (IREPA) as in the following:

$$E_{y,fl} = days\ at\ sea_{y,fl} \cdot \sum_{v=1}^{vessels_{fl}} \left(\frac{days\ in\ service_{v,y,fl}}{days\ in\ year_y} \cdot LOA_{v,fl}^3\right) \qquad \text{Eq. 9}$$

$$E_{y,fl} = days\ at\ sea_{y,fl} \cdot \sum_{v=1}^{vessels_{fl}} \left(\frac{days\ in\ service_{v,y,fl}}{days\ in\ year_y}\right) \qquad \text{Eq. 10}$$

Monthly fishing effort was then calculated also considering the temporary fishing closure for OTB, TBB and PTM in August (as practiced in Italy), the seasonal dynamics for PS and normalized by the value of first year (2005) in order to obtain relative effort changes needed by EwE (Figure 6). A general trend of a reduction of fishery effort was observed in the NEAS area, primarily for otter trawlers (OTB), "lampara" purse seines (PS) and dredges (DRB), while the small-scale fisheries in Slovenia show the most constant trend. During the 2012, the only midwater pair pelagic trawlers (PTM) in Slovenia was decommissioned.

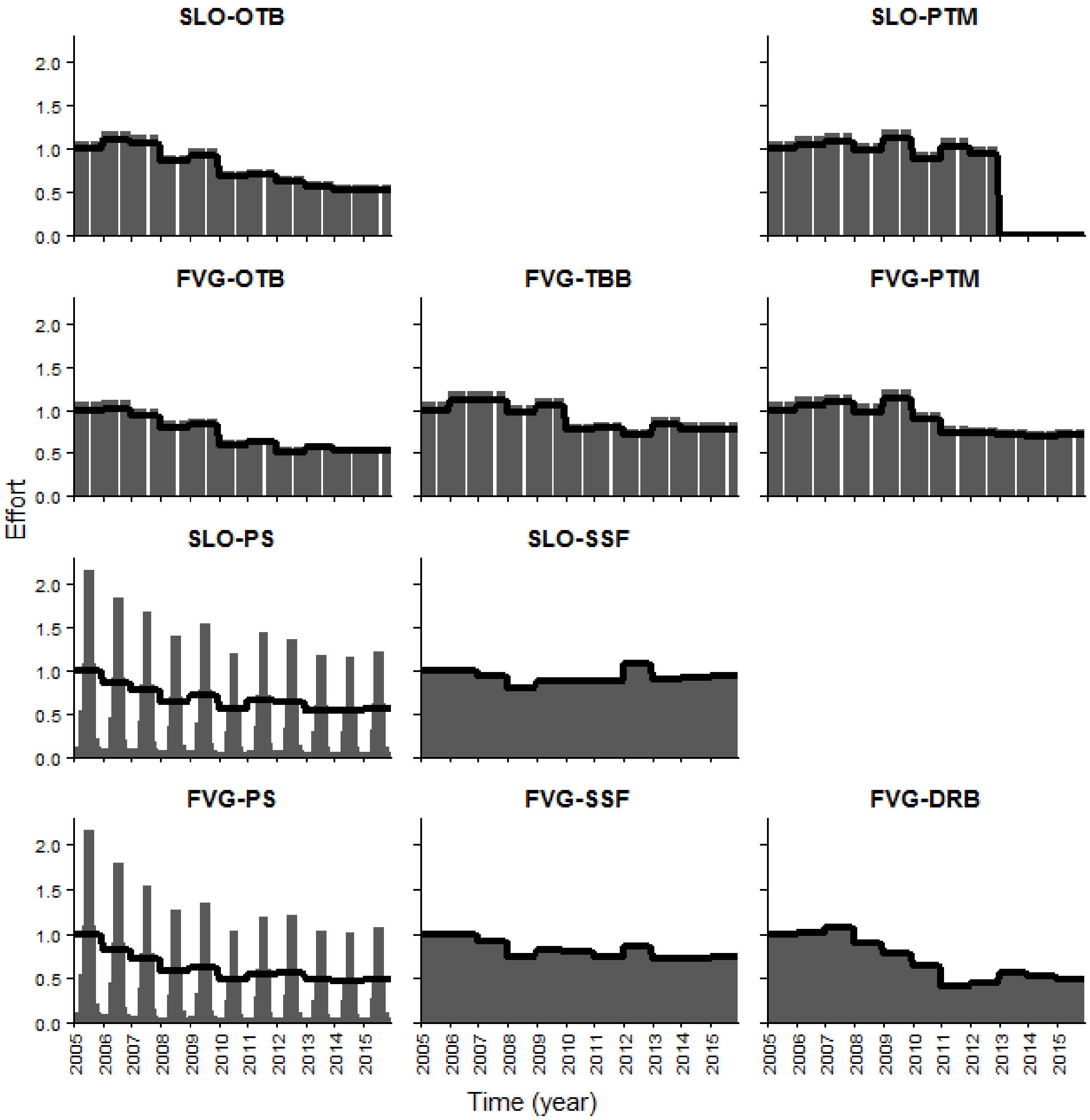

Figure 6. Fishing effort dynamics by gear and by country (Italy's Friuli Venezia Giulia region - FVG, and Slovenia - SLO) relative to the initial effort value in 2005. Monthly effort is represented by bars and includes representation of temporary fleet closures, while the bold line represents the mean yearly relative effort. In future scenarios fishing effort is considered constant to the 2015 value. Fishing gear codes: bottom otter trawl (OTB); "rapido" beam trawl (TBB); midwater pelagic trawl (PTM); "lampara" purse seine (PS); small-scale fisheries (SSF); hydraulic dredges (DRB).

3. The ecosystem model: Ecopath with Ecosim

3.1. The Ecopath with Ecosim modelling suite

The North-East Adriatic Sea (NEAS) ecosystem was modelled using the Ecopath with Ecosim (EwE) suite because of its holistic approach, the standardised structure and the possibility to account both the environmental and fishing effects.

Ecopath with Ecosim (hereafter EwE) is a free general ecosystem model suite with more than 30 years of history (Villasante et al., 2016a). The latest release of EwE, version 6.5 released in July 2016, is composed of three distinct interconnected modelling tools: Ecopath – a module for developing static, mass-balanced snapshot of the system; Ecosim – a time dynamic simulation module for policy exploration; and Ecospace – a spatial and temporal dynamic module. The Ecopath software can be used to: i) address ecological questions, ii) evaluate ecosystem effects of fishing, iii) explore management policies options, iv) evaluate impact and placement of marine protected areas and v) evaluate effects of environmental changes (Heymans et al., 2013). EwE is developed into a user-friendly package, a graphical user interface (GUI) for the Microsoft Windows operating system. EwE is widely adopted for developing aquatic models and food web models in general resulting in more than 400 publications based on models developed with this tool (Colléter et al., 2015). In 2007, EwE was named as one of the ten biggest scientific breakthroughs in NOAA's 200-year history.

The origin of the EwE goes back to 1984 when Jeffry J. Polovina (NOAA) created the mass-balanced model of the food web of a coral reef ecosystem of French Frigate Shoals in the North-western Hawaiian Islands to estimate (Polovina, 1984). Later, the code was implemented into a user-friendly interface by Christensen and Pauly (1992) that integrated theoretical ecology previously developed by R.E. Ulanowicz (Ulanowicz and Puccia, 1990; Ulanowicz, 1986). However, real ecosystems are more complicated than the mass-balance fluxes of biomass as described in Ecopath. In order to include time dynamic changes, impacts of fisheries modifications, as well as environmental drivers, essential to add realism to the approach, in 1995 a time dynamic module, named Ecosim, was added (Walters et al., 1997; Walters et al., 2000). The model was improved by allowing explicit representation of life history stages to represent species that have ontogenetic trophic changes, effects of fishing selectivity and other factors (Walters et al., 1997; Walters et al., 2000; Walters et al., 2008). The shortcoming of the EwE developed so far was the assumption of homogenous spatial behaviour and this was solved by the inclusion of the spatial module Ecospace (Walters et al., 1999). The inclusion of Ecospace in EwE allowed representing, for instance, effects of spatial fisheries management policies, such as definition of no-take zones or marine protected areas (MPA). In contrast to “efficient predator models” as the multispecies virtual population analysis (MSVPA; Giaslason, 1999) who assume the predator is always able to consume its daily food ration, EwE is categorised as “hungry predator model” (Plagányi, 2007). Thus predation in the dynamic modules is based on the foraging arena theory (Walters et al., 1997) where the predator consumption rate of a certain prey is related to that prey’s abundance (Arhens et al., 2012).

Further developments of the EwE allowed the integration of structured data from Global Information System (GIS) (Steenbeek et al., 2013) and the evaluation of ecological indicators with the ECOIND plug-in (Coll and Steenbeek, 2017).

The EwE suite has been developed through time (approximately 35 years) by tackling issues proposed by different applicative case studies, challenges on its various capabilities proposed by users, and observations of inadequacies to represent data; in this way EwE was constantly improved (Christensen and Walters, 2004; Coll et al., 2016). EwE input data and processes rely on single-species assessments that are not meant as substitutes for stock assessments. Its complexity and uncertainty make EwE suitable for strategic/long-term management questions, as opposed to single species assessments, for example, that are more suited for tactical management. In particular, EwE’s advantage is its capability to evaluate the impact of fishing on non-target species, to quantify indirect effects and thus for evaluating trade-offs in mixed fisheries (Walters et al., 2005; Mackinson et al., 2009; Stabler et al., 2016). EwE is also applicable in data-poor situations or in cases with sparse data, not continuous through time or through the food web. Care should be given in defining the correct trophic interactions, as for example an excessive bottom-up control may render the model excessively stable, enabling compensatory responses that lower the risks of overfishing (Christensen et al., 2008). One major disadvantage, common to majority of other ecosystem models, is the low capability to reproduce changes in the structure of the ecosystem, i.e. the models’ structures are fixed and defined during their development phase (Plagányi, 2007). Therefore, although examples of colonisation by invasive species (e.g. Libralato et al., 2015) or extinction of local species (Fortibuoni et al., 2016) are reported in literature, adding or removing functional groups/nodes is not straightforward and easy to reproduce (Plagányi, 2007).

EwE’s source code has been also re-written in FORTRAN (Akoglu et al., 2014; Akoglu et al., 2015), MATLAB (Kearney et al., 2013) and R (Lucey et al., 2014). These reprogramming efforts reproduced only parts of the EwE (the main mass-balance and time-dynamic modelling routines) and allowed advanced uses of the EwE modelling approach, such as end-to-end model development (Akoglu et al., 2015). For instance, the FORTRAN version allows to set-up holistic representations of marine ecosystems by bidirectional coupling of the EwE model with physical and biogeochemical models, which are often written in FORTRAN. Other applications of reprogramming efforts included facilitation to analyse the model's sensitivity and uncertainty, and fast creation of model ensembles and scenarios (Steenbeek et al., 2016).

3.2. The Ecopath mass-balance model

The Ecopath mass balance model can be summarised in two master equations: i) the use of the production of a functional group in the ecosystem and ii) the metabolic balance for the functional group. The first Ecopath master equation (Eq. 11) describes how the production term for each group (i) can be split in components:

$$production = predation\ mortality + catches + \ biomass\ accumulation \\ + net\ migration + other\ mortality \qquad \text{Eq. 11}$$

This mass balance equation assumes that for each functional group there cannot be more biomass lost than it is produced. It can be written more formally as (Eq. 12):

$$B_i \left(\frac{P}{B}\right)_i = \sum_{j=1}^{n} B_j \left(\frac{Q}{B}\right)_j DC_{ji} + B_i \left(\frac{P}{B}\right)_i (1 - EE_i) + \ Y_i + \ E_i + \ BA_i \qquad \text{Eq. 12}$$

where B_i is the biomass of the group i, P/B_i is the production biomass ratio, B_j is the biomass of the predator j of prey i, Q/B_i is the predator's consumption/biomass ratio, DC_{ji} is the fraction of prey i in the diet of predator j, EE_i is the ecotrophic efficiency of group i, i.e. how much of group i's mortality is explained by the model, Y_i is the total fishery catch of group i, E_i is the net migration rate (emigration - immigration) and BA_i is the biomass accumulation rate of group i. In the present model both the net migration rate and the biomass accumulation were left to 0. In a balanced model the EE_i cannot be larger than 1, i.e., there should be a complete balance between uses of energy and matter in the food web, in accordance with the first law of thermodynamics.

The Ecopath mass balance equation is complemented with the functional group's metabolic balance (Eq. 13), which estimates the respiration based on input data about the consumption, production and the non-assimilated food and can be summarised as:

$$consumption = production + respiration + non\_assimilated\ food \qquad \text{Eq. 13}$$

The last assumption is in accordance with the 2nd law of thermodynamics, the irreversibility of natural processes, where to produce some work part of the energy is lost, here represented by respiration. Therefore, the second assumption for Ecopath is that respiration cannot be negative. Respiration is the part of consumption (energy or matter) that is not used for production or recycled as faeces or urine. Here production excludes the primary production. The equation can be expressed also as (Eq. 14):

$$B_i\left(\frac{Q}{B}\right)_i = B_i\left(\frac{P}{B}\right)_i(1-PP) + R_i + B_i\left(\frac{Q}{B}\right)_i na_i \qquad \text{Eq. 14}$$

Where PP_i is the proportion of the total production that can be attributed to primary production, 1 in plants, 0 in heterotrophic consumers and intermediate in e.g. corals, R_i is the group's *i* respiration and na_i is the fraction of food that is not assimilated, i.e. is egested or excreted.

3.3. Ecosim time-dynamic model

Ecosim module in the EwE suite provides a dynamic simulation of the ecosystem, starting from the initial conditions and parameterisation defined in the Ecopath mass-balanced model. The dynamics are expressed by a series of coupled differential equations derived from the Ecopath system of linear equations (Walters et al., 1997; Walters et al., 2000; Christensen and Walters, 2004), where each state variable represents the biomass flux rate of consumer (Eq. 15), primary producer (Eq. 16) and non-living (Eq. 17) functional group, described as follows:

$$\frac{dB_i(t)}{dt} = \gamma_i \cdot \sum_{j=1}^{N} Q_{ji}(t) - \sum_{j=1}^{N} Q_{ij}(t) + I_i - (M_i + e_i)\cdot B_i(t) - \sum_{g=1}^{G}\left[F_{ig}^{m}(t) + F_{ig}^{d}(t)\right]\cdot B_i(t) \qquad \text{Eq. 15}$$

$$\frac{dB_i(t)}{dt} = PP_i(t)\cdot B_i(t) - \sum_{j=1}^{N} Q_{ij}(t) + I_i - (M_i + e_i)\cdot B_i(t) \qquad \text{Eq. 16}$$

$$\frac{dD_i(t)}{dt} = \sum_{j=1}^{N}\left[\delta_{j,i}\cdot\left(M_j\cdot B_j(t) + u_j\sum_{k=1}^{N} Q_{kj}(t)\right)\right] + \sum_{g=1}^{G}\left(\delta_{g,i}\cdot\sum_{j=1}^{N} F_{jg}^{d}(t)\cdot B_j(t)\right) - \sum_{j=1}^{N} Q_{ij}(t) \qquad \text{Eq. 17}$$

In these equations dB_i/d_t is the rate of change of biomass *(B)* of group *i* over time *t*, Υ is the growth efficiency, $\sum Q_{ji}$ is the sum of the consumptions of group *i* over all of its preys *j*, $\sum Q_{ij}$ is the sum of the predation on group *i* by all of its predators *j*, *I* is the immigration, *M* is the non-predatory natural mortality, F is the fisheries mortality induced by each gear g through marketable catches ($C_m = F^m{}_i\ B_i$) and discards ($C_d = F^d{}_i\ B_i$); e is the emigration rate of group i (Walters et al., 1997). PPi represents the primary production rate for autotroph groups. The growth efficiency is represented as $\Upsilon_i=1-(r_i+u_i)$ to explicitly account the respiration rate (r_i) and unassimilation of food (rate, u_i). Respiration flows ($r_i \sum Q_{ji}$) are assumed to be lost, while unassimilated food ($u_i \sum Q_{ji}$), natural mortalities ($M_i\ B_i$) and discards ($F^d{}_i\ B_i$) flow into the opportune detritus compartments, D_i, on the basis of discard fate parameters (δ_{ji} and δ_{gi}) and can be used by the scavengers of the food web (Eq. 17). This system of differential equations is numerically integrated over time under the influence of forcing functions (typically fishing mortalities and/or efforts, changes in primary productivity) starting from the initial condition settings defined by the Ecopath module.

An important characteristic of Ecosim is the ability to explore the implications on system dynamics of different trophic interactions of how the biomass of functional groups in the ecosystem is controlled (Walters et al., 2000). The two extremes of these interactions are the predator control (top-down) and the prey control (bottom-up) (Christensen et al., 2008). Ecosim is capable of modulating a range of these interactions between its two extremes by modification of a set of parameters that allows implementing the “foraging arena” theory (Arhens et al., 2012).

The theory’s basic assertion is that the prey population is partitioned into its vulnerable and invulnerable population components caused by spatial and temporal restrictions in predator and prey activity and the exchange rates between these two components limits the predation rates (Figure 7; adapted from Ahrens et al., 2012). For example, the prey may hide in crevices of coral reefs or inside the school when not feeding, and are subject to predation only when they feed after leaving their shelter. The vulnerability term (v) indicates how severely the increase of predator biomass will affect a given prey’s predation mortality. Trophic interactions in Ecosim are modelled by using the vulnerability parameter, expressing the degree of bottom-up versus the top-down control.

Bottom-up control, i.e. low vulnerability, close to 1, means a variation in predator biomass will not cause noticeable variation on predation mortality of the prey. In this situation, the prey is protected most of the time and becomes available to predators only when it leaves the hideout, thus putting an emphasis on the prey’s behaviour.

On the contrary, top-down control is achieved with high vulnerability values (ex. 10 or higher), meaning that the prey’s predation mortality will tend to increase in a proportional way as the predator biomass increases. Top down control, i.e. Lotka-Voltera dynamics, will manifest as rapid oscillations of prey and predator biomasses.

Generally, top-down control is expected to happen when the predator’s biomass is low, while a shift towards the bottom-up control should be in place when the predator biomass is nearing its carrying capacity (Arhens et al., 2012).

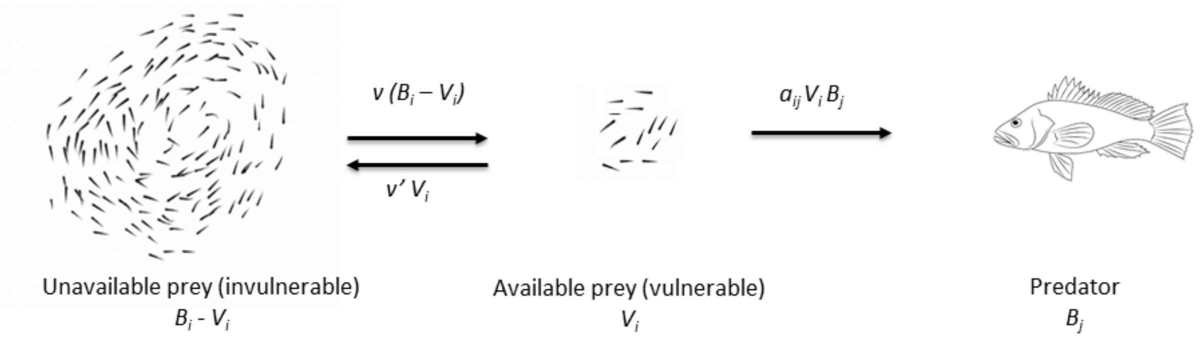

Figure 7. Graphic representation of the foraging arena theory and the prey population split in its invulnerable group (B_i - V_i) and the vulnerable group (V_i) available to the predator (B_j). The vulnerable population gains biomass at rate $v(B_i - V_i)$ and loses it at rates $v'V_i$ and $a_{ij}V_iB_j$. Vulnerability ($v=v'$) and predator search rate (a). Adapted from Walters at al. (1997).

The consumption in Ecosim integrates both the “foraging arena” theory (vulnerabilities) as well as other trophic parameters, forcing functions and mediation functions (Eq. 18 and 19), thus describing the temporal dynamics of trophic interaction between functional groups in the ecosystem, affecting biomasses as per the Ecosim equations (Eq. 15, 16 and 17). Formally the consumption equations are expressed as:

$$Q_{i,j} = \frac{a_{i,j} \cdot v_{i,j} \cdot B_i \cdot B_j \cdot T_i \cdot T_j \cdot S_{i,j} \cdot M_{i,j}/D_j}{v_{i,j} + v_{i,j} \cdot T_i \cdot M_{i,j} + a_{i,j} \cdot B_j \cdot T_j \cdot S_{i,j} \cdot M_{i,j}/D_j} \qquad \text{Eq. 18}$$

$$D_j = \frac{h_j \cdot T_j}{1 + \sum_i (a_{i,j}) \cdot B_i \cdot T_i \cdot M_{i,j}} \qquad \text{Eq. 19}$$

Where $Q_{i,j}$ is the consumption of prey i by predator j, $v_{i,j}$is the vulnerability of prey i to predator j, $B_i$and $B_j$the biomass of the prey i and the predator j. Other trophic parameters are represented by the predator’s j search rate for prey i ($a_{i,j}$, prey and predator relative feeding time (T_i and T_j), effects of handling time as a limit to consumption rate (D_j) and the predator handling time (h_j). Forcing functions defined by the user are the seasonal or long-term forcing function ($S_{i,j}$) and the mediation effect forcing function ($M_{i,j}$).

Although vulnerability is among the most important parameters in an Ecosim model and is used for tuning the model around available data, there are other parameters that can influence ecosystem dynamics, in particular the handling time and the switching behaviour parameter (see Walters et al., 1997; 2000; Christensen et al., 2008; Ahrens et al., 2012).

Handling time effect represents the notion that predators split their time between searching for prey and other activities; such as pursuit, manipulation and ingestion; here grouped together as the time needed to "handle" the prey (Christensen et al., 2008). In Ecosim this parameter limits the maximum consumption rate a functional group can have, and is represented as a ratio between the maximum and the Ecopath base consumption rate. Such limit triggers primarily in situations at low predator density (or high prey density) and when a vulnerable prey density increases greatly, so without such a limit predator's productivity unrealistically or too optimistic (Christensen et al., 2008).

Table 4. List of parameters that influence the model fitting and their description.

Parameter	Description
Vulnerability (*v*)	Expresses the trophic interaction between a prey and a predator, i.e. indicates how severely the increase of predator biomass will affect a given prey's predation mortality from bottom-up to top-down control.
Maximum relative production rate	It limits the production rates of primary producers.
Handling time (h_j, *Group info in Ecosim*)	Handling time assumes that predators have a limited time for foraging, used for "handling the preys" (pursuit, manipulation or ingestion) rather than searching for prey. It is set by limiting the maximal consumption rate over the initial consumption rate
Maximum feeding time (*Group info in Ecosim*)	Maximum feeding time limits how much a predator can increase the time spent for feeding and exposing themselves to an increasing predation risk.
Satiation (*Feeding time adjust rate, Group info in Ecosim*)	Feeding time adjust rate represents the adjustment speed of the foraging time in order for predators to maintain their consumption rates near the initial base rate in the Ecopath model.
Predator effect on feeding time (*Group info in Ecosim*)	This parameter works in conjunction with the Feeding time adjust rate, and regulates if and how much the target food consumption rate, the time exposed to predation risk, will be reduced in case the predator abundance increases.
Fraction of other mortality sensitive to changes in feeding time (*Group info in Ecosim*)	This parameter is also related to the feeding time and determines what proportion of the natural mortality $(P/B)_i(1-EE_i)$ varies proportionally to the relative time spent feeding. As density increases feeding time usually has to increase to maintain food consumption rate, and this increased feeding time leads to higher mortality rates, resulting in a density dependent natural mortality.
Density dependent catchability (*Group info in Ecosim*)	Density dependent catchability parameter allows to correct the catchability values when this change in relation to the population size or the inhabited area. Particularly, when the biomasses decrease, the species may behave by aggregating in smaller areas but still subject to the same fishing pressure and consequently caught.
Switching power parameter (*Group info in Ecosim*)	The switching behaviour indicates that predators change the proportions of each prey in their diet more rapidly than so does the relative biomass of that prey in the environment. For example, the predators may stop searching for a rare prey or alternatively increase disproportionally the proportion of prey that has an increase in biomass.

The switching behaviour in Ecosim governs how the proportions of the predator's diet should change at a variation in predator abundance in the environment. The predator is said to have a "switching behaviour" if it takes disproportionally more (or less) of a certain prey as it becomes more (or less) abundant. This parameter ranges from 0 to 2, and describes behaviours as: a) no switching behaviour (parameter = 0), b) a prey must become very rare before the predator stops searching for them (parameter <1) and c) predator switches violently when predator increases or decreases (parameter > 1, opportunistic feeding behaviour).

A brief description of all the parameters governing the Ecosim model is listed in Table 4.

3.4. Fitting an Ecosim model to observations.

The basic requirement for any credible model for policy analysis is the ability to reproduce at least the observed historical trends and to reproduce responses to disturbances such as fishing.

For that purpose, Ecosim compares the simulation data with the observed data, biomass and/or catch time series. Each reference time series can be assigned a weight, based on how variable or reliable those data are. Ecosim measures the goodness of fit between model outputs and observations as the weighted sum of squared deviations of log observed biomasses/catch from log predicted biomasses/catch data (Eq. 20) and summing them (Eq. 21). In case of relative biomass/catch data, the predicted biomass/catch data are scaled by the maximum likelihood estimate of the relative biomass/catch scaling factor *q* in the equation (Eq. 22).

$$SS_{D_{fg}} = \sum \left[log_e D_{pred_{fg}} - log_e D_{obs_{fg}} \right]^2 \qquad \text{Eq. 20}$$

$$SS_{fg} = SS_{B_{fg}} + SS_{C_{fg}} \qquad \text{Eq. 21}$$

$$relD_{obs} = q \cdot D_{pred} \qquad \text{Eq. 22}$$

Where $D_{pred_{fg}}$, $D_{obs_{fg}}$ and $relD_{obs}$ are the predicted, observed and relative biomass or catch data. The *SS* for functional group (SS_{fg}) is the sum of biomass SS ($SS_{B_{fg}}$) and catch SS ($SS_{C_{fg}}$).

Reference data may not cover all time steps of the simulation. On the other hand, complete data for all time steps are required for any forcing using in the model, such as the forced biomass, the fishing effort, the fishing mortality or other forcing and mediating functions (Christensen et al., 2008).

Fitting a model to time series reference data is a step-wise iterative process by correcting the parameter estimates and time series forcing data to find what values could reproduce more closely the observed historical patterns. In such process improving the fit for one functional group may improve or degrade the model fits for other groups due to trophic links between them. In the same way the overall model fit, measured by the SS, may improve but the fit of some functional groups to observed data may be worse.

In Ecosim there are two main fitting procedures that can be adopted: a) a search for relative primary production values, b) search for vulnerability estimates.

The first method searches for a time series of values of relative primary production that may represent historical productivity regime shifts impacting the biomasses throughout the ecosystem and this way improve the fit to observed data. However, while in fact this method may improve the fit and give useful information, it is only limited to the hindcast period. In some cases, it is used after fitting obtained with vulnerabilities optimization to improve the fitness.

Vulnerabilities are estimated by a nonlinear optimisation routine that tries to minimize the model’s measure of goodness (SS). A sensitivity analysis can identify those vulnerabilities with the major influence to the model simulation, and reduce the number of vulnerabilities to most relevant ones (i.e. the number of parameters) to be estimated. Such analysis is done by slightly changing each vulnerability and rerunning the model and measure how much the SS has changed. It is not recommendable to fit a large ecosystem model using one big nonlinear estimation scheme that simultaneously varies all the uncertain model parameters (Christensen et al., 2008).

3.5. Strategies for fitting the Ecosim model to observations

In ecological modelling, typically there is not a single best-for-all solution for representing functional groups in complex food webs. The historical trends of some functional groups will be better reproduced than others. This is caused by the inherent complexity and variability of the ecosystem, by the limitation of data in synthesizing the main ecosystem dynamics, by the role of main driving processes not always being clearly defined, by the assumptions embedded in modelling necessarily simplifying the ecosystem main actors, drivers and processes. Thus, in a series of fitting there might be different outcomes of parameter optimisation procedures because of different directions the optimisation takes in a multi-dimensional space in search for the minimum values given the individual simulation settings.

The uncertainty of observations, thus on input data and model parameterisation, therefore, is a factor contributing to the difficulties in representing the system. Therefore, any simulation even a well fitted one is not completely right or completely wrong, but instead it represents some dynamic characteristic of the modelled ecosystem. An ensemble of simulations may address the variability issue observed in nature by focusing on different possible scenarios or model settings in each simulation. Such ensembles may also be useful for comparing model results with the uncertainties of the observations.

In the Ecopath with Ecosim environment, the ensembles can be obtained by varying i) the initial conditions and parameters in the mass-balanced Ecopath and ii) by varying time-dynamic interaction parameters in the Ecosim module. Both procedures may provide a framework for evaluating the model’s uncertainty (Figure 8). In the first case the Monte Carlo analysis changes the initial conditions and parameters around their mean values and creates a set of new mass-balanced Ecopath models that share the same Ecosim settings obtained through the fitting procedure. Alternatively, in the second case the ensemble is based on the same mass-balanced Ecopath model while variations are made in the Ecosim model. Some of those variations may be: a) different Group info parameter settings, b) selection of different reference data, c) different vulnerability values and different selection of vulnerability parameters to be estimated, d) variation of the forcing functions and e) different environmental parameters. Combinations of the mentioned variations may also provide different model properties.

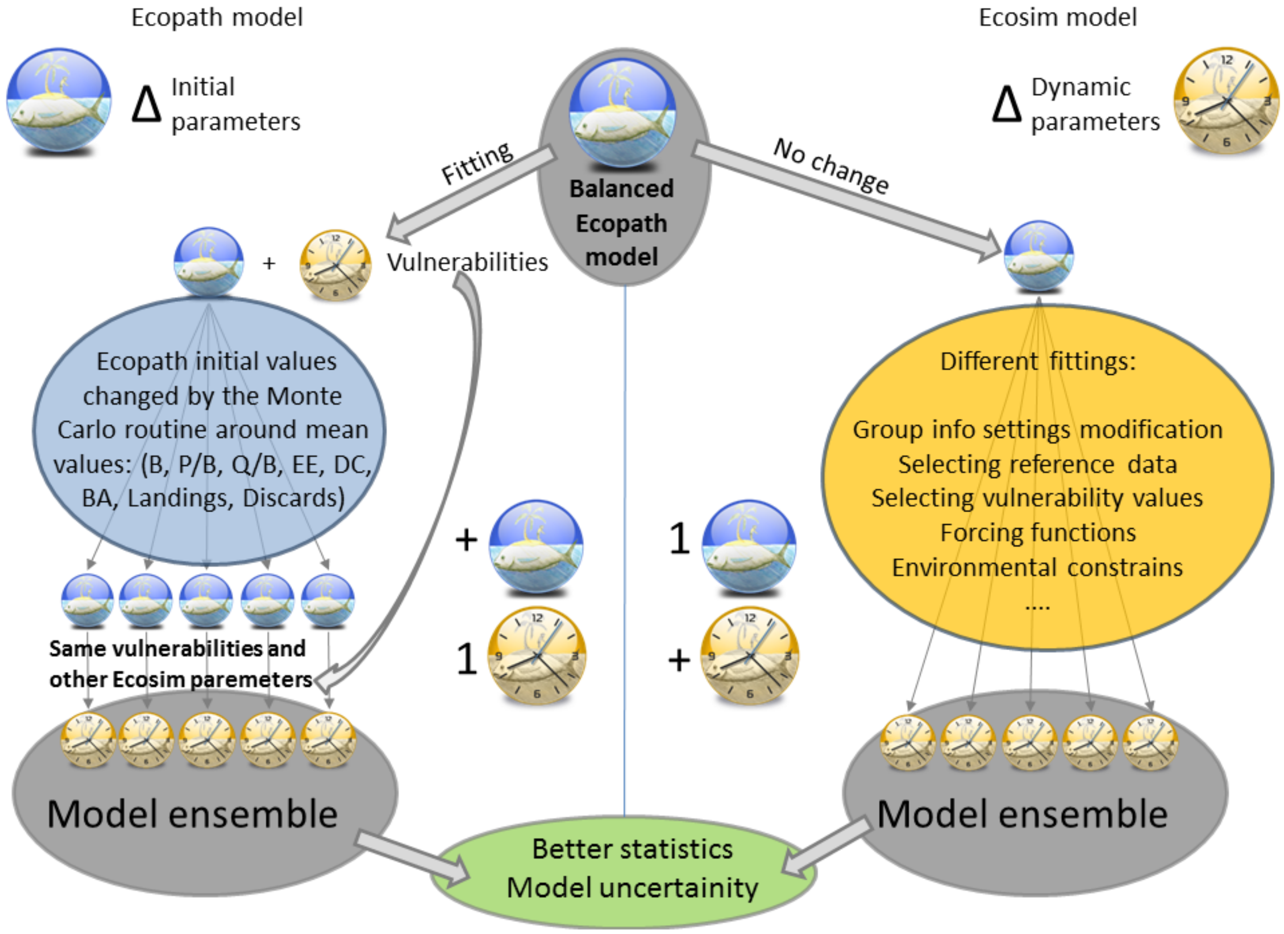

Figure 8. Schematic representation of possible options for creating an ensemble of models and assessing the model's uncertainty. On the left, an ensemble of models can be obtained by changing the Ecopath initial parameters upon which are applied the same Ecosim parameters and vulnerabilities obtained during the initial fitting of the model to observed data. On the right, an ensemble of model can be obtained by maintaining the same Ecopath mass-balanced model and instead change the Ecosim parameters, forcing functions or environmental parameters.

4. *Ecopath model: the ecosystem of the North-East Adriatic Sea (NEAS)*

4.1. Structure of the NEAS model

The North-east Adriatic Sea model resulted from expanding, updating and improving existing models developed for the area (Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010), by reviewing the functional groups, including new available data for calibrating the model with time series of biomass and catches. The EwE model suite belongs to the so called "aggregated models", because of their propriety to simplify the ecosystem by aggregating several species into a functional group. Functional groups may be represented by individual species or groups of species that perform a similar function in the ecosystem. Species composing a functional group should occupy similar ecological niches and show similar biomass and catch trends over time; i.e. they should meet the criteria of having approximately similar growth rates, consumption rates, diets, habitats and predators.

The model comprises 33 functional groups: 3 primary producers, 27 consumers and 3 detritus groups. Consumers are represented by: Marine mammals (FG 1), Marine birds (FG 2), Marine turtles (FG 3), Elasmobranchii (sharks and rays; FG 4), 10 fish groups (FGs 5-14), Cephalopods (FG 15), Mussel farms and Bivalves (FGs 16, 17), 6 groups for benthic invertebrates (FGs 18-23), heterotrophic plankton (FGs 24-27), while autotrophic plankton (FGs 28, 29), Macroalgae & seagrasses (FG 30) are the primary producer groups. To simulate the use of the organic matter in the pelagic food web and to account for fishery's discard the detritus group is split in three parts: Particulate organic matter (POM; FG 31), Fishery discard (FG 32) and Bottom detritus (FG 33) (Figure 9, Table 5, 6). Figure 4 illustrates the transfer of energy from low trophic levels (detritus groups and primary producers), highlighting the importance planktonic loop and the benthic filter feeders that transfer the biomass to higher trophic levels. Most of the flow of biomass in NEAS is also concentrated in the planktonic loop and some benthic filter feeders like Bivalvia, Suprabenthos and Annelida & other worms.

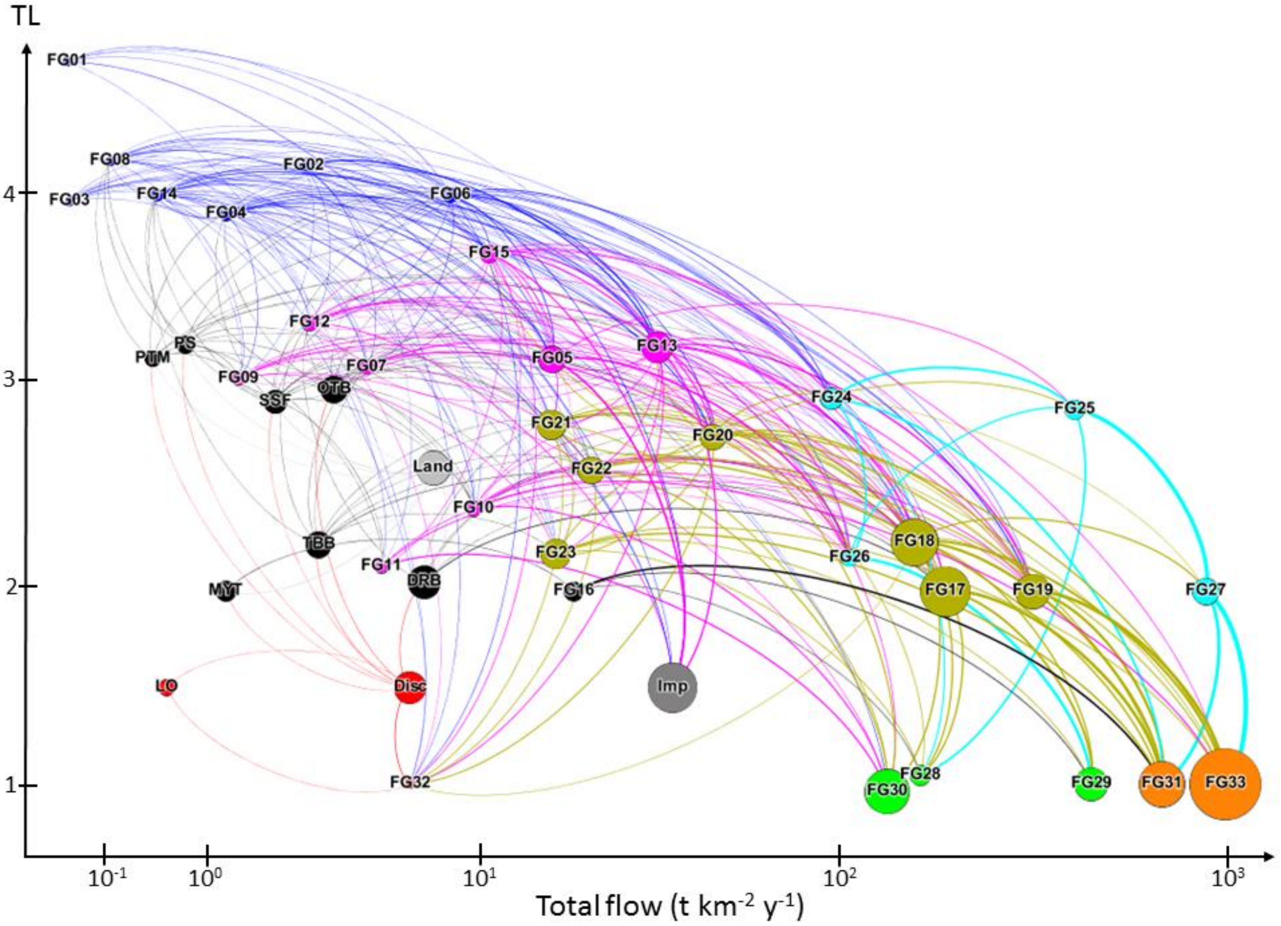

Figure 9. Scheme of the NEAS food web model developed in Ecopath with Ecosim: circles represent functional groups and main fishing gear. Circle sizes are proportional to log-normalized biomass for functional groups and to catch for fishing fleets. Functional groups grouped by colour: higher predators (blue), intermediate level mobile consumers (pink), benthic macrofauna (dark yellow), planktonic loop (light blue), primary producers (green), POM and bottom detritus (orange), discards (red), imported diet (*Imp*, grey). Fishing fleets are in black colour and the marketable landings (*Land*) are in grey. The biomass fluxes are represented by the curved lines, the fluxes direction is counter-clockwise, line colour is according to the target endpoint's colour and line thickness is proportional to flow intensity. Functional group names are available in Table 6; fisheries: otter trawlers (*OTB*), rapido trawl (*TBB*), mid-water pair pelagic trawl (*PTM*), purse seines (*PS*), small-scale fisheries (*SSF*), dredges (*DRB*) and mussel aquaculture (*MYT*); discards: LO subject discard (*LO*), other discards (*Disc*).

The present NEAS model adopts wet weight biomass (t km^{-2}) as the model's currency unit, while the biomasses of each functional group biomasses are based on 2005. The analysis of Ecopath results will, thus, allow to have insights into the situation of the ecosystem in 2005 (snapshot image of the ecosystem state; Chapter 4), and the year 2005 will represent the initial conditions for the dynamic simulation with Ecosim (Chapter 5 and 6).

4.2. Input data

Details on the biomass of each functional group and the parameters used in the model are summarised in Table 6. The biomass of Marine mammals was derived from census activities conducted in the NEAS area (Genov et al., 2008), while the census for the local nesting population was used for Marine bird biomass (Zenatello et al., 2014). The inshore fish community was estimated form visual census surveys along the coastal shore areas (Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca and Lipej, 2005), also accounting for the coverage of each habitat type. The offshore fish and benthic communities were estimated from the two trawl surveys available in the area: the MEDITS (Bertrand et al., 2002) and SoleMON (Grati et al., 2013). Small pelagic fish biomass was estimated form acoustic survey data (Leonori, 2006); mussel farms were expressed as shell-free wet weight as in Solidoro et al. (2010). The biomass of seagrass was derived from the maps of seagrass habitat distribution and average density (Peduzzi and Vukovič, 1990; Turk and Orlando Bonaca, 2002; Curiel et al., 2005; EMODnet, www.emodnet-seabedhabitats.eu).

Table 5. Ecopath parameter values and Ecosim functional group info parameters used in the NEAS model. Values in bold are initial Ecopath values and parameters set in Ecosim. (P/B: Production, Q/B: Consumption, EE: Ecotrophic efficiency, P/Q: Production/consumption ratio, F/Z: fishing mortality, Unass.cons: Unassimilated consumption. * Consumption derived from clearance data, thus unassimilation include the pseudo-faeces as estimated in the NEAS by Solidoro et al., 2010. Abbreviated codes: as in Table 6.

	Group code	Trophic level	Biomass (t/km^2)	P/B (/year)	Q/B (/year)	EE	P/Q	Unass. cons.	F/Z	proportion of discards in diet	Omnivory index
1	MarMamm	4.65	**0.0150**	**0.08**	**11.01**	0.0000	0.007	**0.200**	0.0000	**1.10%**	2.047
2	MarBird	4.19	**0.0529**	**4.61**	**69.34**	0.0000	0.066	**0.200**	0.0000	**1.00%**	0.062
3	MarTurt	4.00	**0.0317**	**0.16**	**2.54**	0.0000	0.065	**0.200**	0.0000	**0.00%**	0.088
4	Elasm	3.91	**0.4386**	**0.31**	**3.95**	0.9939	0.078	**0.200**	0.8297	**0.00%**	0.332
5	SmallPel	3.20	**5.0000**	**1.90**	**9.13**	0.6212	0.208	**0.200**	0.1979	**0.00%**	0.618
6	BenthPel	3.99	0.7646	**1.70**	**5.82**	**0.9000**	0.292	**0.200**	0.7448	**3.00%**	0.305
7	CoastPla	3.13	**0.6377**	**1.07**	**8.40**	0.8211	0.127	**0.200**	0.1104	**0.00%**	0.004
8	PelPisc	4.21	**0.0815**	**0.57**	**5.13**	0.3159	0.111	**0.200**	0.1805	**0.00%**	0.139
9	InvFeed	3.02	**0.3376**	0.87	**5.80**	0.8368	**0.150**	**0.300**	0.5152	**0.00%**	0.169
10	Detritiv	2.39	**0.6236**	**0.91**	**17.70**	0.6117	0.051	**0.500**	0.3777	**0.00%**	0.319
11	Herbiv	2.11	**0.4156**	**0.99**	**14.40**	0.4209	0.068	**0.400**	0.0030	**0.00%**	0.105
12	Flatfish	3.33	**0.8021**	**1.43**	**6.13**	0.3503	0.233	**0.200**	0.1826	**1.00%**	0.397
13	Benthiv	3.20	**6.5292**	**2.45**	**6.70**	0.6740	0.365	**0.200**	0.0723	**3.50%**	0.348
14	DemPisc	4.01	**0.2860**	**1.00**	**5.24**	0.3238	0.190	**0.200**	0.1309	**1.00%**	0.377
15	Cephal	3.71	**1.3438**	**3.10**	**12.97**	0.5640	0.239	**0.200**	0.1657	**3.00%**	0.729
16	MusselF	2.00	**1.5386**	**1.99**	**13.59**	0.5685	0.146	**0.775***	0.5685	**0.00%**	0.000
17	Bivalv	2.00	**42.0000**	**0.70**	4.66	0.8627	**0.150**	**0.650**	0.2230	**0.00%**	0.000
18	AnnWorm	2.05	**30.9370**	**0.80**	5.37	0.7032	**0.150**	**0.260**	0.0006	**0.15%**	0.047
19	Supraben	2.00	8.2800	**4.67**	**35.43**	**0.7000**	0.131	**0.250**	0.0000	**0.00%**	0.000
20	DecaSto	2.75	**3.5000**	**4.30**	**14.00**	0.9529	0.307	**0.200**	0.0465	**5.93%**	0.528
21	Gastrop	2.84	**5.5000**	**1.06**	**3.13**	0.7779	0.338	**0.300**	0.0636	**5.50%**	0.146
22	Echinod	2.11	**4.0072**	**0.84**	5.63	0.9507	**0.150**	**0.300**	0.1590	**4.59%**	0.157
23	OthBenth	2.19	**5.8221**	**1.06**	**3.13**	0.7234	0.338	**0.200**	0.0754	**0.00%**	0.339
24	MacroZoo	2.99	**2.0000**	**14.60**	**50.48**	0.1948	0.289	**0.200**	$3*10^{-07}$	**0.00%**	0.770
25	MicroZoo	2.94	**1.7070**	**177.80**	**254.00**	0.1749	0.700	**0.165**	0.0000	**0.00%**	0.054
26	MesoZoo	2.17	**1.0480**	**61.80**	**107.40**	0.7380	0.575	**0.124**	0.0000	**0.00%**	0.304
27	BactPla	2.00	**3.8890**	**141.66**	**244.35**	0.7536	0.579	**0.185**	0.0000	**0.00%**	0.000
28	Phy1Dino	1.00	**1.7641**	**92.03**		0.3087			0.0000		
29	Phy2Diat	1.00	**7.8371**	**61.19**		0.3120			0.0000		
30	AlgSeagr	1.00	**24.2500**	**6.13**		0.1910			0.0000		
31	POM	1.00	**26.7168**			0.8102					0.429
32	FishDisc	1.00	**0.0001**			0.9951					0.000
33	BottDetr	1.00	**296.2990**			0.9971					0.714

Table 6. Input data by functional group for the North-east Adriatic Sea model and relative reference.

1. MarMamm		Marine mammals
Species		*Tursiops truncatus*
Biomass (t km^{-2})	0.0150	Genov et al., 2008
P/B (y^{-1})	0.08	Coll et al., 2007
Q/B (y^{-1})	11.01	Coll et al., 2007
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010 and modifications

2. MarBird		Marine birds
Species		*Phalocrocorax carbo, Casmerodius albus, Ardea cinerea, Chroicocephalus ridibundus, Larus melanocephalus, Larus canus, Larus argentatus*
Biomass (t km^{-2})	0.0529	Zenatello et al., 2014
P/B (y^{-1})	4.61	Coll et al., 2007
Q/B (y^{-1})	69.34	Coll et al., 2007
Diet		Cosolo et al., 2010; Cosolo et al., 2011

3. MarTurt		Marine turtles
Species		*Caretta caretta*
Biomass (t km^{-2})	0.0317	Coll et al., 2007
P/B (y^{-1})	0.16	Coll et al., 2007
Q/B (y^{-1})	2.54	Coll et al., 2007
Diet		Tomas et al., 2001

4. Elasm		Elasmobranchii
Species		*Dasyatis pastinaca, Mustelus mustelus, Mustelus punctulatus, Myliobatis aquila, Pteroplatytrygon violacea, Raja asterias, Raja clavata, Scyliorhinus stellaris, Squalus acanthias, Torpedo torpedo*
Biomass (t km^{-2})	0.4386	MEDITS
P/B (y^{-1})	0.31	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	3.95	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Ellis et al., 1996; Capapé et al., 2007; Saïdi et al., 2009; Valls et al., 2011; Navarro et al., 2013

5. SmallPel		Small pelagic fish
Species		*Engraulis encrasicolus, Sardina pilchardus, Sprattus sprattus*
Biomass (t km^{-2})	5.0000	Leonori, 2005
P/B (y^{-1})	1.90	GFCM, 2012
Q/B (y^{-1})	9.13	Coll et al., 2007
Diet		Borme et al., 2009; Nikolioudakis et al., 2012; Borme et al., 2013; Legovini, 2013

6. BenthPel		Benthopelagic fish
Species		*Boops boops, Merlangius merlangus, Scomber japonicus, Scomber scombrus, Trachurus mediterraneus, Trachurus trachurus, Trisopterus capelanus*
Biomass (t km^{-2})		Estimated by Ecopath
P/B (y^{-1})	1.70	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	5.82	Empirical equation from Pauly et al., 1990
EE	0.90	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Šantić et al., 2004; Milić, 2012

7. CoastPla		Coastal planktivorous fish
Species		*Alosa fallax, Aphia minuta, Atherina spp., Chromis chromis*
Biomass (t km^{-2})	0.6377	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005
P/B (y^{-1})	1.07	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	8.40	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Bartulović et al., 2004

8. PelPisc		Pelagic piscivorous fish
Species		*Auxis rochei, Dicentrarchus labrax, Euthynnus alletteratus, Lichia amia*
Biomass (t km^{-2})	0.0815	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005
P/B (y^{-1})	0.57	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	5.13	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Coetzee, 1982; Falautano et al., 2007

9. InvFeed		Invertebrate feeding fish
Species		*Diplodus annularis, Diplodus sargus sargus, Diplodus vulgaris, Spondyliosoma cantharus, Sparus aurata*
Biomass (t km^{-2})	0.3376	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005
P/B (y^{-1})		Estimated by Ecopath
Q/B (y^{-1})	5.80	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/Q	0.15	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Sabatini, 2008

Table 6. Input data and references by functional group for the North-east Adriatic Sea model. (continuation)

10. Detritiv		Detritivorous fish
Species		Mugilidae
Biomass (t km^{-2})	0.6236	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005
P/B (y^{-1})	0.91	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	17.70	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010 and modifications

11. Herbiv		Herbivorous fish
Species		*Sarpa salpa*
Biomass (t km^{-2})	0.4156	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005
P/B (y^{-1})	0.99	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	14.40	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010

12. Flatfish		Flatfish
Species		*Arnoglossus laterna, Arnoglossus thori, Citharus linguatula, Platichthys flesus, Buglossidium luteum, Monochirus hispidus, Pegusa lascaris, Solea aegyptiaca, Solea solea, Pegusa impar, Synapturichthys kleinii, Microchirus variegatus*
Biomass (t km^{-2})	0.8021	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005; SoleMON survey
P/B (y^{-1})	1.43	From stock assessment of *Solea solea*
Q/B (y^{-1})	6.13	Empirical equation from Pauly et al., 1990
Diet		Fanelli et al., 2009

13. Benthiv		Benthivorous fish
Species		Blenniidae, Callionymidae, Cepolidae, Gobiidae, Syngnathidae, Triglidae, *Conger conger, Lithognathus mormyrus, Mullus barbatus, Mullus surmuletus, Pagellus acarne, Pagellus bogaraveo, Pagellus erythrinus, Scorpaena notate, Serranus hepatus, Spicara spp., Umbrina cirrosa*
Biomass (t km^{-2})	6.5292	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005; SoleMON and MEDITS survey
P/B (y^{-1})	2.45	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	6.70	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Fabi et al., 2006; Fanelli et al., 2010; Stagioni et al., 2012

14. DemPisc		Demersal piscivorous fish
Species		*Merluccius merluccius, Trachinus draco, Uranoscopus scaber, Scophthalmus maximus, Scophthalmus rhombus, Scorpaena porcus, Scorpaena scrofa, Zeus faber*
Biomass (t km^{-2})	0.2860	Lipej et al., 2003; Guidetti et al., 2005; Orlando Bonaca et al., 2005; SoleMON and MEDITS survey
P/B (y^{-1})	1.00	Coll et al., 2007
Q/B (y^{-1})	5.24	Empirical equation from Pauly et al., 1990
Diet		Stagioni et al., 2013

15. Cephal		Cephalopoda
Species		*Alloteuthis media, Eledone moschata, Loligo vulgaris, Sepia elegans, Sepia officinalis, Sepiola spp.*
Biomass (t km^{-2})	0.0815	SoleMON and MEDITS survey
P/B (y^{-1})	0.57	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	5.13	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Castro and Guerra, 1990; Coelho et al., 1997; Krstulović Šifner and Vrgoč, 2009; Neves et al., 2009

16. MusselF		Mussel farms
Species		*Mytilus galloprovincialis* farms
Biomass (t km^{-2})	1.3438	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010, corrected to include Slovenia
P/B (y^{-1})	3.10	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	12.97	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010

17. Bivalv		Bivalvia
Species		*Acanthocardia tuberculate, Aequipecten opercularis, Atrina fragilis, Callista chione, Corbula gibba, Flexopecten glaber, Pecten jacobaeus*
Biomass (t km^{-2})	42.0000	SoleMON survey and unpublished data from Callista chione survey
P/B (y^{-1})	0.70	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/Q	0.15	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010

18. AnnWorm		Annelida & other worms
Species		*Aphrodita aculeate, Aspidosiphon (Aspidosiphon) muelleri muelleri* and other Errantia and similar phyla
Biomass (t km^{-2})	30.9370	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/B (y^{-1})	0.80	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/Q	0.15	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010

Table 6. Input data and references by functional group for the North-east Adriatic Sea model. (continuation)

19. Supraben		Suprabenthos
Species		Peracarida
Biomass (t km^{-2})		Estimated by Ecopath
P/B (y^{-1})	4.67	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	35.43	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
EE	0.70	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Navarro-Barranco et al., 2013; Guerra-García et al., 2014
20. DecaSto		Decapoda & Stomatopoda
Species		Anomura, Astacidea, Brachyura, Caridea, Penaeoidea, *Squilla mantis*, and other Decapoda
Biomass (t km^{-2})	3.5000	SoleMON survey
P/B (y^{-1})	4.30	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	14.00	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Froglia and Giannini, 1989; Freire, 1996; Bernárdez et al., 2000; Oh et al., 2001; Janas and Baranska, 2008
21. Gastrop		Gastropoda
Species		*Aporrhais pespelecani, Bolinus brandaris, Hexaplex trunculus, Turritella communis* and other Gastropoda
Biomass (t km^{-2})	5.5000	SoleMON survey
P/B (y^{-1})	1.06	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	3.13	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
22. Echinod		Echinodermata
Species		*Astropecten irregularis, Holothuria (Holothuria) tubulosa, Leptopentacta spp., Ocnus planci, Ophiothrix fragilis, Ophiura ophiura, Paracentrotus lividus, Psammechinus microtuberculatus, Sphaerechinus granularis* and other Echinodermata
Biomass (t km^{-2})	4.0072	SoleMON survey
P/B (y^{-1})	0.84	Libralato et al. 2006, Libralato et al. 2010, Solidoro et al. 2010
P/Q	0.15	Libralato et al. 2006, Libralato et al. 2010, Solidoro et al. 2010
Diet		Bonsdorff and Vahl, 1982; Dearborn et al., 1991; Loh and Todd, 2011
23. OthBenth		Other benthic filter feeders
Species		*Botryllus schlosseri, Calliactis parasitica, Microcosmus vulgaris, Molgula spp., Phallusia mammillata, Polycarpa spp., Pyura spp., Suberites domuncula, Tethya aurantium*, Cnidaria, Porifera, Tunicata and other sessile fauna
Biomass (t km^{-2})	5.8221	SoleMON survey
P/B (y^{-1})	1.06	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	3.13	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
24. MacroZoo		Macrozooplankton & jellyfish
Species		
Biomass (t km^{-2})	2.0000	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/B (y^{-1})	14.60	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	50.48	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
25. MicroZoo		Microzooplankton
Species		
Biomass (t km^{-2})	1.7070	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010; Lipitzer et al., 2015
P/B (y^{-1})	177.80	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	254.00	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Unassim. cons.	0.165	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
26. MesoZoo		Mesozooplankton
Species		
Biomass (t km^{-2})	1.0480	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010; Mozetič et al., 2012
P/B (y^{-1})	61.8	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	107.40	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Unassim. cons.	0.124	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Diet		Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
27. BactPla		Bacterioplankton
Species		Bacteria
Biomass (t km^{-2})	3.8890	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/B (y^{-1})	141.66	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Q/B (y^{-1})	244.35	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
Unassim. cons.	0.185	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010

Table 6. Input data and references by functional group for the North-east Adriatic Sea model. (continuation)

28. Phy1Dino		Phytoplankton 1 – Dinoflagellate
Species		Dinoflagellate
Biomass (t km^{-2})	1.7641	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/B (y^{-1})	92.03	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
29. Phy2Diat		Phytoplankton 2 – Diatoms
Species		Diatoms
Biomass (t km^{-2})	7.8371	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
P/B (y^{-1})	61.19	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
30. AlgSeagr		Macroalgae & seagrass
Species		*Cymodocea nodosa, Zoostera marina, Zoostera noltii, Posidonia oceanica, Cystoseira spp., Fucus spp.*
Biomass (t km^{-2})	24.2500	Peduzzi and Vukovič, 1990; Turk and Orlando Bonaca, 2002; Curiel et al., 2005; EMODnet maps
P/B (y^{-1})	6.13	Peduzzi and Vukovič, 1990.
31. POM		Particulated organic matter
Biomass (t km^{-2})	26.7168	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010
32. FishDisc		Fishery discard
Biomass (t km^{-2})	0.0001	Practically continuously consumed and inexistent according to Bozzano and Sardà (2002)
33. BottDetr		Bottom detritus
Biomass (t km^{-2})	296.2990	Libralato et al., 2006; Libralato et al., 2010; Solidoro et al., 2010

The diet matrix was updated for all the consumers, except for the zooplankton, Bacterioplankton and shellfish groups. Diet data were preferably for the study area, the Adriatic Sea or, if unavailable, from the nearest area or with similar characteristics as the NEAS (Table 7). Migratory functional groups had part of their consumed biomass imported from outside the modelled area, i.e. Marine mammals (60%), Small pelagic fish (50%), Benthopelagic fish (10%), Pelagic piscivorous fish (10%), Flatfish (20%), Bentivorous fish (20%), Demersal piscivorous fish (10%) and Cephalopoda (30%). The proportion of fishery discard in the diet of functional groups with scavenging behaviour or known discard consumers (Kaiser and Spencer, 1994; Hall-Spencer et al., 1999; Bozzano and Sardà, 2002) was set to be large enough to equal the flow of discards on the basis of observed evidence of no discard accumulation in the system (Bozzano and Sardà, 2002).

Table 7. Diet matrix of the NEAS model. Preys are listed on the left and predators on the top; the table is rotated anticlockwise.

	prey/predat	1	2	3	4	5	6	7	8	9	10	11	12	13
1	MarMamm													
2	MarBird													
3	MarTurt													
4	Elasm						0.0050							
5	SmallPel	0.0800	0.0113	0.0687	0.0937		0.1360		0.6024				0.0048	0.0109
6	BenthPel	0.2300	0.0176	0.0200	0.0136				0.0121					
7	CoastPla		0.0142		0.0100		0.0164		0.0149					0.0026
8	PelPisc				0.0036									
9	InvFeed		0.0016		0.0274				0.0101					
10	Detritiv		0.0016		0.0020				0.0100					
11	Herbiv		0.0007											
12	Flatfish		0.0061		0.0058		0.0117		0.0108					0.0013
13	Benthiv	0.0390	0.9352	0.1235	0.1993		0.3402		0.1074				0.1155	
14	DemPisc		0.0018	0.0033	0.0278									
15	Cephal	0.0400		0.0139	0.1151		0.0405		0.0696				0.0182	0.0081
16	MusselF													
17	Bivalv			0.0006	0.0064		0.0107			0.4035	0.0100		0.0122	0.0547
18	AnnWorm			0.0132	0.0310		0.0107	0.1163	0.0027	0.1697	0.1400	0.0500	0.1072	0.1707
19	Supraben			0.0045	0.1626		0.0192	0.1147	0.0015	0.0100	0.1400	0.0500	0.3888	0.2379
20	DecaSto			0.0243	0.2602		0.0919		0.0094	0.0840	0.0300		0.1031	0.1503
21	Gastrop			0.0089	0.0318		0.0012		0.0014	0.0200			0.0004	0.0115
22	Echinod			0.0029	0.0014					0.0796				0.0155
23	OthBenth			0.0443	0.0036				0.0038	0.1223				0.0143
24	MacroZoo			0.6717	0.0002		0.1663		0.0439		0.0150		0.0147	0.0362
25	MicroZoo					0.0200								
26	MesoZoo				0.0044	0.4800	0.0200	0.7691	0.0002	0.0100	0.0150	0.0100	0.0251	0.0378
27	BactPla													
28	Phy1Dino													
29	Phy2Diat													
30	AlgSeagr								0.0000	0.1009	0.3000	0.8900		0.0132
31	POM													
32	FishDisc	0.0110	0.0100				0.0300						0.0100	0.0350
33	BottDetr										0.3500			
	Import	0.6000				0.5000	0.1000		0.1000				0.2000	0.2000

Table 7. Diet matrix of the NEAS model. (continuation)

	prey/predat	14	15	16	17	18	19	20	21	22	23	24	25	26	27
1	MarMamm														
2	MarBird														
3	MarTurt														
4	Elasm	0.0001													
5	SmallPel	0.0931	0.0940					0.0135							
6	BenthPel	0.0100	0.0031												
7	CoastPla	0.0231	0.0107												
8	PelPisc														
9	InvFeed	0.0100	0.0013												
10	Detritiv	0.0100	0.0060												
11	Herbiv	0.0100	0.0089												
12	Flatfish	0.0203	0.0010												
13	Benthiv	0.4739	0.1190					0.0188							
14	DemPisc														
15	Cephal	0.0579						0.0146							
16	MusselF														
17	Bivalv		0.0006					0.0484	0.7011	0.0418					
18	AnnWorm	0.0002	0.0106					0.1134	0.0511						
19	Supraben	0.0621	0.0270			0.0100		0.1893	0.0190	0.0033					
20	DecaSto	0.0399	0.2946												
21	Gastrop	0.0018	0.0245					0.0469		0.0369					
22	Echinod	0.0003						0.0341	0.0100						
23	OthBenth		0.0001					0.0496	0.0400						
24	MacroZoo	0.0470	0.0375					0.0475							
25	MicroZoo										0.1000	0.4000		0.0887	
26	MesoZoo	0.0023	0.0002					0.0095				0.1900			
27	BactPla					0.0400			0.0095				0.9419		
28	Phy1Dino			0.0065	0.0500	0.0500		0.0148			0.0500	0.0500	0.0581		
29	Phy2Diat			0.0288	0.1900	0.0500					0.0500			0.9113	
30	AlgSeagr	0.0280				0.0500		0.0945	0.0803	0.2043					
31	POM			0.9647	0.7600	0.5000	0.2850	0.0738		0.5359	0.7000	0.3600			0.2000
32	FishDisc	0.0100	0.0300			0.0015		0.0593	0.0550	0.0459					
33	BottDetr		0.0310			0.2985	0.7150	0.1720	0.0340	0.1318	0.1000				0.8000
	Import	0.1000	0.3000												

Productivity rates (P/B) were updated for the Macroalgae & seagrass group from a study conducted locally in Slovenia (Peduzzi and Vukovič, 1990) and for the Small pelagic fish from stock assessment data (GFCM, 2012). For Benthopelagic fish, Demersal piscivorous fish and Flatfish the consumption rates (Q/B) were updated according to Palomares and Pauly's equation (1989) (Eq. 23):

$$\log Q/B = 7.964 - 0.204\, W_\infty - 1.965\, T' + 0.083\, A + 0.532\, h + 0.398\, d \qquad \text{Eq. 23}$$

Where W_∞ is the asymptotic weight, T' is the temperature parameter expressed as T' = 1000/mean annual temperature in Kelvin, A is the aspect ratio of the caudal fin, h is a dummy variable expressing food type (1 for herbivores and 0 for detritivoures and carnivores) and d is a dummy variable also for food type (1 for detritivoures, 0 for herbivores and carnivores). Other parameters were maintained as in the original model (Libralato et al., 2006; Libralato et al., 2010).

The multi-gear fishery of the NEAS domain was represented through six operational units: bottom otter trawl (OTB), "*rapido*" trawl (TBB), midwater pair pelagic trawl (PTM), light aided purse seine "*lampara*" (PS), mollusc dredge (DRB) and small scale fisheries (SSF). Exploitation activities included in the model were detailed by gear and country, including one mussel farming activity and 10 fishing fleets (Italy: OTB, TBB, PTM, PS, SSF and DRB and Slovenia: OTB, PTM, PS and SSF). Slovenia's reconstructed landings data (Bolje et al., 2014), were already disaggregated by gear and by species. Italian official data collected by IREPA (2005-2012, www.irepa.org/it), instead, needed to be disaggregated into operational fleet units, where "other fish species" was disaggregated by using the combined landings of Trieste's fish market and Slovenia's landing data for species not explicitly reported in IREPA and resulted in a disaggregation into Benthopelagic fish, Coastal planktivorous fish, Pelagic piscivorous fish, Invertebrate feeding fish, Flatfish, Benthivorous fish and Demersal piscivorous fish groups. Italian landing data for OTB, TBB, PTM and DRB were corrected to 70%, 80%, 50% and 90% respectively as these fleets exert part of their activity outside the study area, while the landings of other fleets were assumed as coming exclusively from the study area (Table 8). Mussel farm landings (ACQ) were expressed as shell-free wet weight as in Solidoro et al. (2010).

Table 8. Landings and dead discards by gear, i.e. caught discards multiplied by mortality rates. Mortalities applied to functional groups by combining different sources form similar gears: 1. 25% - Rodriguez-Cabello et al. (2005), 2. 100% assumed mortality, 3. 90% - Berghahn et al. (1992), Broadhurst et al., (2006) with corrections, 4. 100% - present assumption and Berghahn (1990), 5. 90% - van Beek et al. (1990) with corrections, 6. 100% - Hill and Wassenberg (1990), 7. 50% - Gasper et al. (2001), Hall-Spencer et al. (1999), 8. 100% - Hall-Spencer et al. (1999), 9. 50% - Gasper et al. (2001), Giomi et al. (2008), Pranovi et al. (2001), Hall-Spencer et al. (1999), 10. 10% - Mensink et al. (2000), Gasper et al. (2001), Hall-Spencer et al. (1999), 11. 40% - Currie and Parry (1999), Kaiser and Spencer (2005), Gasper et al. (2001), Bergman and Moore (2001), Pranovi et al. (2001), Hall-Spencer et al. (1999), 12. 50% - Wassenberg and Hill (1993), Currie and Parry (1999), Hall-Spencer et al. (1999).

	Landings (L_{tot}, t)							Dead discards (D_{tot}, t)						
	OTB	TBB	PTM	PS	SSF	DRB	ACQ	OTB	TBB	PTM	PS	SSF	DRB	ACQ
Elasm [1]	27.9	0.2	6.31		78.5			0.2						
SmallPel [2]	5.2		715.5	835.3	6.7			172.7		73.4	71.0			
BentPel [3]	126.1		32.1	69.9	94.7			617.4		0.7	1.8	24.7		
CoastPla [4]	19.3			5.1	9.4			41.3	0.1					
PelPisc			0.5	0.1	7.6									
InvFeed [2]	13.3		0.1	1.1	100.6			7.0				28.9		
Detritiv [2]	13.8		10.7	17.9	150.7			0.1		20.9				
Herbiv				0.8	0.4									
Flatfish [5]	1.2	48.2			151.3			2.1	1.7				4.7	
Benthiv [2]	192.1		14.7	30.7	400.5			475.3		1.59	0.9	40.4		
DemPis [2]	14.2	4.0		2.0	16.4			0.7						
Cephal [6]	249.5	51.2		5.5	348.7			24.5	10.8					
MusselF [2]							1586.8							158.6
Bivalv [7]	0.01	252.4			537.7	1599.8		0.01	1154.2				3014.2	
AnnWorm [8]								7.3	7.4				0.3	
DecaSto [9]	63.9	14.5			232.2			28.2	84.1			167.3	110.7	
Gastrop [10]	21.5	172.4			73.3			39.7	31.9			3.5	28.5	
Echinod [11]								473.2	62.9	0.5			1.8	
OthBenth [12]								335.5	129.8					
MacroZoo [2]								0.01						

Discards were estimated from Fabi and Grati (2005) for the small-scale fishery, Santojanni (2005) for “lampara” purse seine and Raicevich (2008) for the other fishing gears and from unpublished data (SOSPECO project) (Table 8). In particular, discard data from the SOSPECO project was extracted at the highest possible detail, i.e. first at the species level then at the functional group level. Mortality of discarded species was assumed to be 100% for the majority of fish species because of damage during the hauls, prolonged air exposure, induced mortality by chilling in ice, or damage during the sorting activities (Hall-Spencer et al., 1999; Hill and Wassenberg, 1990). Available mortalities from literature for similar gears and species were considered for horse mackerels (*Trachurus spp.*), flatfish, invertebrates, sharks and rays (Table 8). In these cases, the dead part of discards flows into the fishery discard functional group, while the alive discarded proportion returns to take part of the biomass of the respective functional groups (Table 8).

Landing prices for each functional group were estimated as the mean price from 2005 until 2012 from IREPA reports.

4.3. Description of the NEAS model

According to the official statistics the total landings in the NEAS in 2005 were 8437 t and the biggest contribution was given by the small-scale fishery (SSF) and the clam hydraulic dredges (DRB), followed by the mussel aquaculture (Figure 10). On the other hand, three fishing gears were responsible for the majority of the 7464 t of discards, namely the hydraulic dredges, the otter trawl and the rapido trawl.

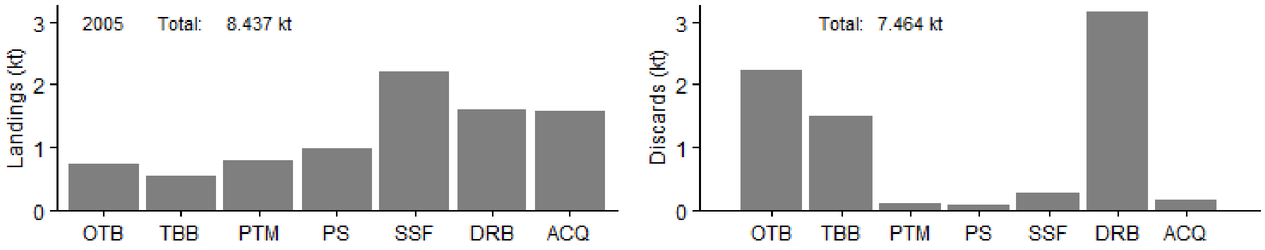

Figure 10. Landings and discards in the NEAS area by fishing gear in 2005. Fishing gears: otter trawlers (OTB), rapido trawl (TBB), mid-water pair pelagic trawl (PTM), purse seines (PS), small-scale fisheries (SSF), dredges (DRB) and mussel farms shell-free wet weight (ACQ)

The omnivory index corrected to exclude the imports in the diet of each predator (Table 7) indicates that the most specialized consumers are the two shellfish groups (Bivalvia and Mussel farms) and the detritus feeders (Bacterioplankton and Suprabenthos), followed by Coastal planktivorous fish, Small pelagic fish, Annelida & other worms and Pelagic piscivorous fish, while the most omnivorous are Cephalopoda, Decapoda & Stomatopoda and Macrozooplankton & jellyfish. Possible impacts of direct and indirect interactions in the NEAS model identified by the Mixed Trophic Impact (MTI) analysis are shown in Figure 11. Among the most positive interactions are those of the Macroalgae & seagrass on the Herbivorous fish, the Benthivorous fish on the Marine birds, and Macrozooplankton & jellyfish on Marine turtles. Negative interactions are those of the Cephalopoda on the Herbivorous fish, Elasmobranchii on both Demersal piscivorous fish and Pelagic piscivorous fish and the effect of Decapoda & Stomatopoda on the benthic fauna (Gastropoda, Echinodermata and Other benthic filter feeders). Also, some possible indirect effects are those of the Gastropoda on the hydraulic dredge fishery (DRB) since Gastropoda have a negative effect on the Bivalvia, the major target of the mentioned fishery.

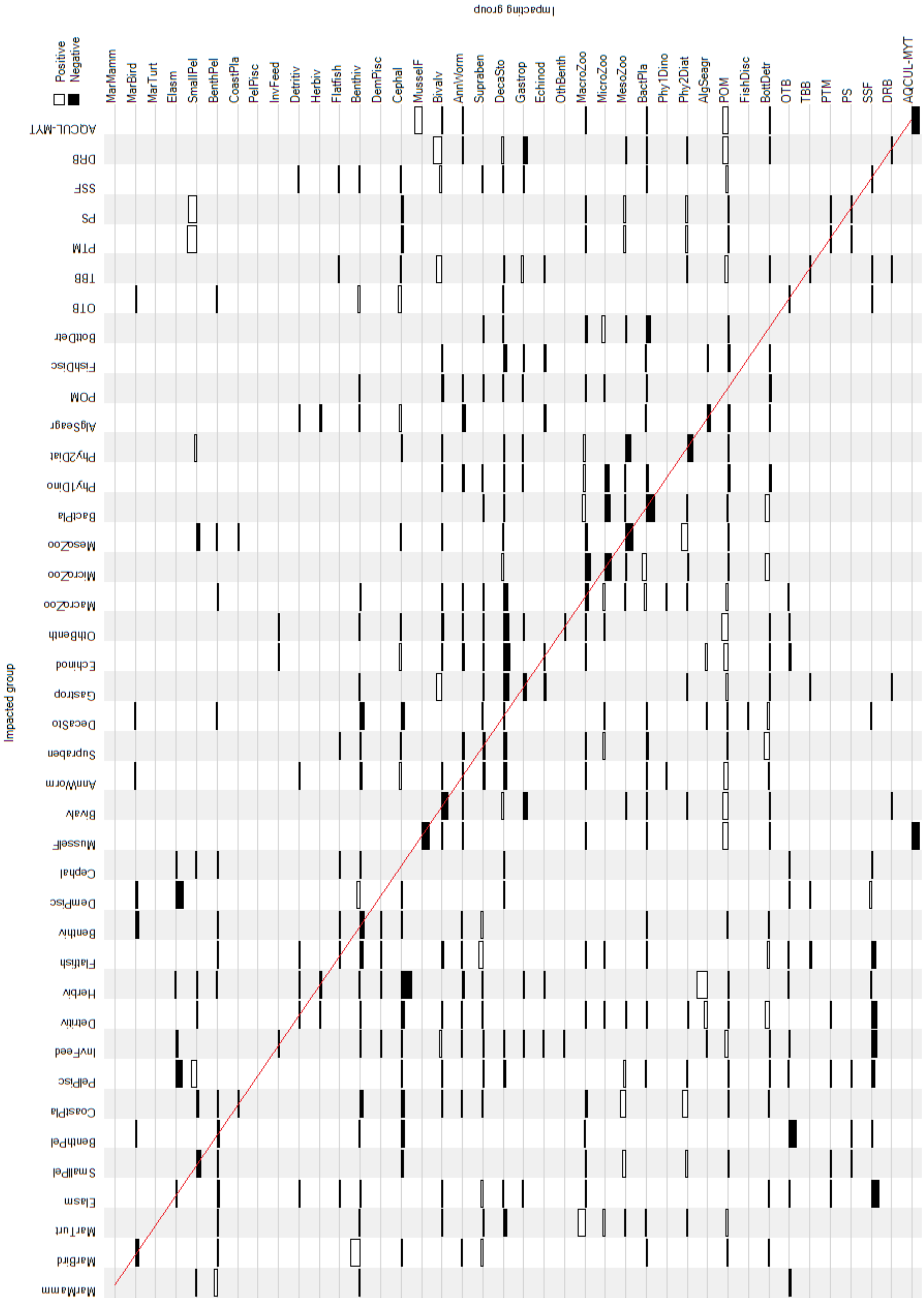

Figure 11. Mixed Trophic Impact analysis (MTI). The bars indicate the relative impact [-1,1], positive impacts are in white and negative ones are in black. The diagonal red line indicates the functional group's impact on itself. The figure is rotated anticlockwise.

The role of each functional group in the food web was estimated using the keystoness index proposed by Libralato et al. (2006). The three benthic groups, the Cephalopoda, Benthivorous fish, and Decapoda & Stomatopoda, had the highest values of keystoness (Figure 12, 13). Within the top ten ranked functional groups are also two primary producers, both with considerable biomasses, (Macroalgae & seagrass and Diatoms) as well as three heterotrophic plankton groups (Macro-, Meso- and Micro-zooplankton), followed by the Bacterioplankton ranked 11th. Ranked at the 6th place is the Elasmobranchii group, with the lowest biomass among the 10 highest ranked functional groups, indicating an important ecological role.

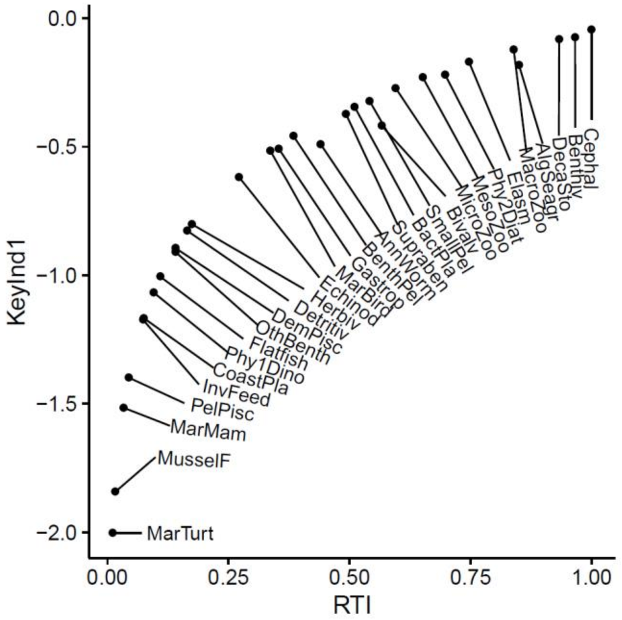

Figure 12. Functional groups ranked by the keystoness index (Libralato et al. 2006) and the relative trophic interaction (RTI) index.

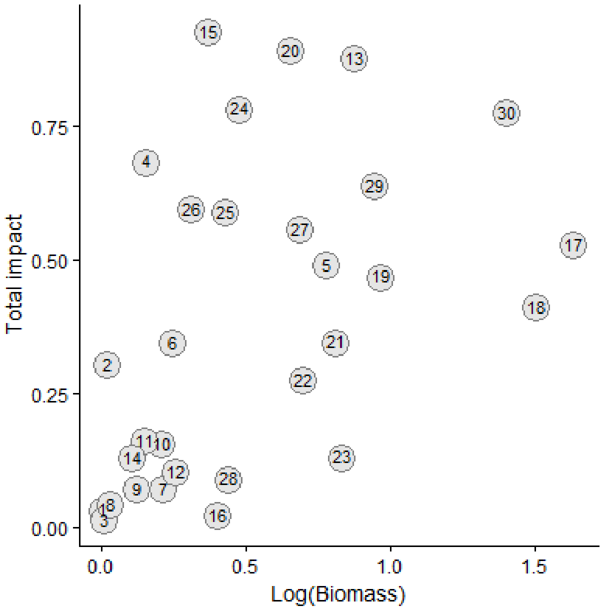

Figure 13. Absolute total impact against log biomass. No fishing is included. The number indicates the functional groups, as in Table 6.

The present model has been also characterised using a set of standardised ecological indicators (Heymans et al., 2014) (Table 9). The comparison of present model with other two Ecopath models in different areas of the Adriatic Sea (Coll et al., 2006; Barausse et al., 2008) shows certain similarities: the export per total system throughput unit is much lower and the biomass per surface unit (detritus excluded) are similar (Barausse et al., 2008). On the contrary, the net primary production per total system throughput unit is lower than both alternative models, an indication of a lower primary production. In addition, the variance of trophic levels in the diets (system omnivory index) and indications of system maturity (i.e. Finn's cycling index and Finn's mean path length) are higher than in the other models, indicating functional groups have a better trophic connection in the NEAS model. Moreover, as shown in Figure 14, the detritus cycle has a predominant role in the present food web, where the fluxes of mass are approximately 8 times greater than the primary production cycle fluxes.

Table 9. Ecological indicators of community energetics, community structure, cycling of nutrients and information theory.

Flows indices	Value	Unit
Sum of all consumption	2529.0 *49.2*	$t\ km^{-2}\ y^{-1}$ *% of TST*
Sum of all exports	15.9 *0.3*	$t\ km^{-2}\ y^{-1}$ *% of TST*
Sum of all respiratory flows	817.7 *15.9*	$t\ km^{-2}\ y^{-1}$ *% of TST*
Sum of all flows into detritus	1777.3 *34.6*	$t\ km^{-2}\ y^{-1}$ *% of TST*
Total system throughput (TST)	5140.0	$t\ km^{-2}\ y^{-1}$
Sum of all production	1900.2	$t\ km^{-2}\ y^{-1}$
Calculated total net primary production	790.6 *15.4*	$t\ km^{-2}\ y^{-1}$ *% of TST*
Total primary production/total respiration	0.966	
Net system production	-27.1	$t\ km^{-2}\ y^{-1}$
Total primary production/total biomass	4.897	
Total biomass/total throughput	0.031	y^{-1}
Total biomass (excluding detritus)	161.4	$t\ km^{-2}$

Network flow indices	Value	Unit
Throughput cycled (excluding detritus)	0.95	$t\ km^{-2}\ y^{-1}$
Predatory cycling index	0.05	% of throughput without detritus
Throughput cycled (including detritus)	1388.00	$t\ km^{-2}\ y^{-1}$
Finn's cycling index	27.04	% of TST
Finn's mean path length	6.15	
System Omnivory Index	0.25	

Information indices	Value	Unit
Ascendency	29.59	%
Overhead	70.41	%
Capacity	27018	flowbits

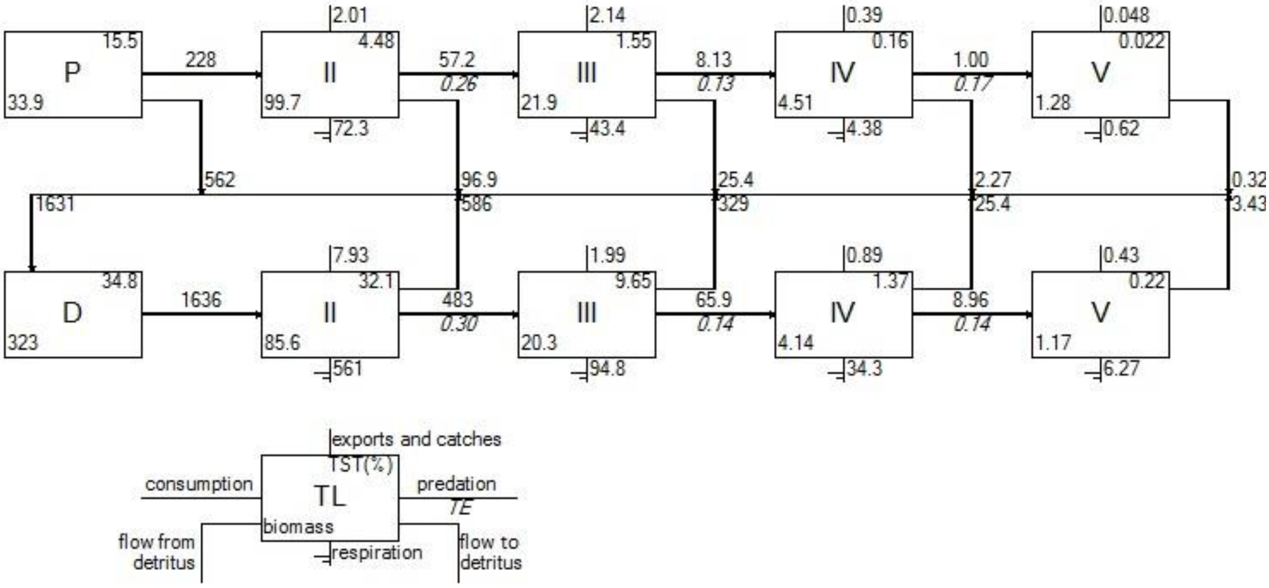

Figure 14. Lindeman Spine representation of the trophic flows from the NEAS area. Roman numbers stand for integrated trophic levels, P for primary producers, D for detritus, TST% for total system throughput, TE for trophic efficiency. Flows are in $t\ km^{-2}\ year^{-1}$ and biomasses $t\ kg^{-2}$.

5. *Performance analysis of the time–dynamic model*

5.1. Fitting scheme for the NEAS model

In the present study, all simulations shared the same mass-balanced Ecopath model, described in Chapter 4 while all Ecosim simulations shared the same time steps, observation data, Group info settings as well as the primary production and consumption forcing functions.

The simulations in Ecosim were run from 2003 to 2030: years 2003-2004 were used as a spin up (using 2005 parameters), 2005-2015 used for fitting with available data (hindcast period), and 2016-2030 for representing future scenarios. The reference model used four forcing functions (production rates for Diatoms, Dinoflagellates, Macroalgae & seagrass and consumption rates for Bacterioplankton) and fishing effort specific by gear. The fishing effort was based on cubic LOA for OTB, TBB and PTM, and number of vessels for PS, DRB and SSF (see details in Chapter 2.7). From 2016 to 2030 the yearly fishing effort was kept constant at the value in 2015 and the monthly forcing functions of the last four years (2012-2015) (Figure 3) used to define climatological values for the future (2016-2030). Fitting was assessed on the basis of biomass time series obtained from MEDITS and SoleMON surveys and yearly catch data (landings and discards) for each fishing gear and functional group. Mussel farms' biomass and landings were set constant by forcing the biomass to a 2005's values, assuming there are no yearly variations.

Ecosim Group info parameters (Table 10) were defined as in Mackinson and Daskalov (2007). Feeding time adjust rates were set to 0.10 for top predators, assuming a low risk for predation mortality for these groups and therefore their feeding time should tend to be constant. Low feeding time adjust rates were set also for sessile benthonic species and some other benthic neritic species, owing to their inability to shelter from predators and therefore not changing substantially their feeding time. Intermediate values were assigned to Marine birds, mid-trophic level predators and planktivorous fish. Feeding time adjust rates of 1 were assigned to zooplankton, Suprabenthos and to Annelida & other worms, allowing these groups to have a high risk of mortality and good trophic connections with their predators. The fraction of other mortality sensitive to changes in feeding time was set to 0 for bivalves and benthic filter feeders assuming they are unable to shelter themselves and have a constant feeding behaviour, and left to 1 for other functional groups. By adjusting the predator effect on feeding time to 0 for plankton, benthic filter feeders and bivalves it is assumed that their feeding is not affected by predator presence. The opposite was done for higher predators (Elasmobranchii, Pelagic piscivorous fish and Cephalopoda) as well as Flatfish, Herbivorous and Detritivorous fish, setting their values to 1. Other fish groups and mobile benthic macro-fauna had intermediate values (0.50 and 0.75). As suggested by Mackinson and Daskalov (2007) these modifications to feeding time should dampen the instabilities in model simulations. In the present model, no density dependant catchability corrections were used. The handling time (h_j, QB_{max}/QB_0) was set to 3 as suggested in Christensen et al. (2008), except for Pelagic piscivorous fish, Detritivorous fish and Flatfish where it was set to 10 to reduce the unrealistic spike effects. These values are much lower than the default value of 1000. Christensen et al. (2008) reported that for most fish bioenergetics models the feeding rates in the natural environment are often low compared to their potential maximum and that typical ratios between these two rates are around 0.3-0.4. Finally, by enabling the switching power parameter for opportunistic diet of higher trophic predators, mainly the Marine birds, allowed to stabilize their highly oscillatory behaviour and unrealistic increase in biomass of certain functional groups as observed during the initial fitting procedure.

For the purpose of these analyses a reference model was created by fitting the Ecopath to observed data. The fitting was carried out iteratively by estimating the vulnerability values. In the first iteration, the 40 most influential vulnerabilities were estimated, and subsequently the number of vulnerabilities increased as no further improvement was observed, until all available vulnerabilities were estimated. By doing so, the use of a massive nonlinear estimation scheme was avoided and the optimisation routine was allowed to initially move in the direction of major influences and only later to adjust for minor variation.

The integration of available biomass and catch time series (in total 243 observations), the forcings of observed fishing effort trends and forcing functions of primary production rates into an unfitted Ecosim model (all vulnerabilities = 2) produced an initial goodness of fit in terms of sum of squared residuals (SS, Eq. 20, 21 and 22) of 107.62, implying a mean absolute error of about 60%. The estimation procedure of vulnerabilities lowered the goodness of fit to 87.43. The later model was used as a reference model (REF) in all further analysis (Chapter 5 and 8), except in Chapters 6 and 7 (where the ensemble of models was used). The model predictions discretly reproduced the trends of the relative observed biomasses (Figure 15) and absolute landings (Figure 16). Among simulations of predicted biomasses, Elasmobranchii, Flatfish and Echinodermata, but also Microzooplankton, had a good agreement with the observed trends (Figure 15). Instead, for simulations of predicted landings, Detritivorous fish, Flatfish and Benthivorous fish were close to the observed trend of absolute landings (Figure 16)

Table 10. Ecosim Group info parameters used in the NEAS model. F.O.M.S.C.F.T.: Fraction of other mortality sensible to changes in feeding time. Abbreviations as in Table 6.

Group code	Max. relative P/B	Max. relative feeding time [0,2]	Feeding time adjust rate [0,1]	F.O.M.S.C.F.T. [0,1]	Predator effect on feeding time [0,1]	Density dependant catchability: Q_{max}/Q_o	QB_{max}/QB_o	Switching power [0,2]
MarMamm		1.25	0.10	1	0.00	1	3	2.0
MarBird		1.25	0.50	1	0.00	1	3	2.0
MarTurt		1.25	0.10	1	0.00	1	3	2.0
Elasm		1.25	0.10	1	1.00	1	3	1.0
SmallPel		1.25	0.50	1	0.50	1	3	0.0
BenthPel		1.25	0.50	1	1.00	1	3	2.0
CoastPla		1.25	0.50	1	0.50	1	3	0.0
PelPisc		1.25	0.10	1	1.00	1	10	0.0
InvFeed		1.25	0.10	1	0.50	1	3	2.0
Detritiv		1.25	0.25	1	1.00	1	10	0.5
Herbiv		1.25	0.10	1	1.00	1	3	0.5
Flatfish		1.25	0.50	1	1.00	1	10	0.0
Benthiv		1.25	0.50	1	0.75	1	3	1.0
DemPisc		1.25	0.10	1	1.00	1	3	1.0
Cephal		1.25	0.10	1	1.00	1	3	1.0
MusselF		1.25	0.10	1	0.00	1	3	0.0
Bivalv		1.00	0.05	0	0.00	1	3	0.0
AnnWorm		1.50	1.00	1	0.50	1	3	0.0
Supraben		1.50	1.00	1	0.50	1	3	0.0
DecaSto		1.25	0.10	1	0.50	1	3	1.0
Gastrop		1.25	0.10	1	0.50	1	3	0.5
Echinod		1.00	0.10	1	0.50	1	3	0.0
OthBenth		1.00	0.05	0	0.00	1	3	0.0
MacroZoo		1.50	1.00	1	0.00	1	3	1.0
MicroZoo		1.50	1.00	1	0.00	1	3	0.0
MesoZoo		1.50	1.00	1	0.00	1	3	0.0
BactPla		1.50	0.05	1	0.00	1	3	0.0
Phy1Dino	2							
Phy2Diat	2							
AlgSeagr	2							

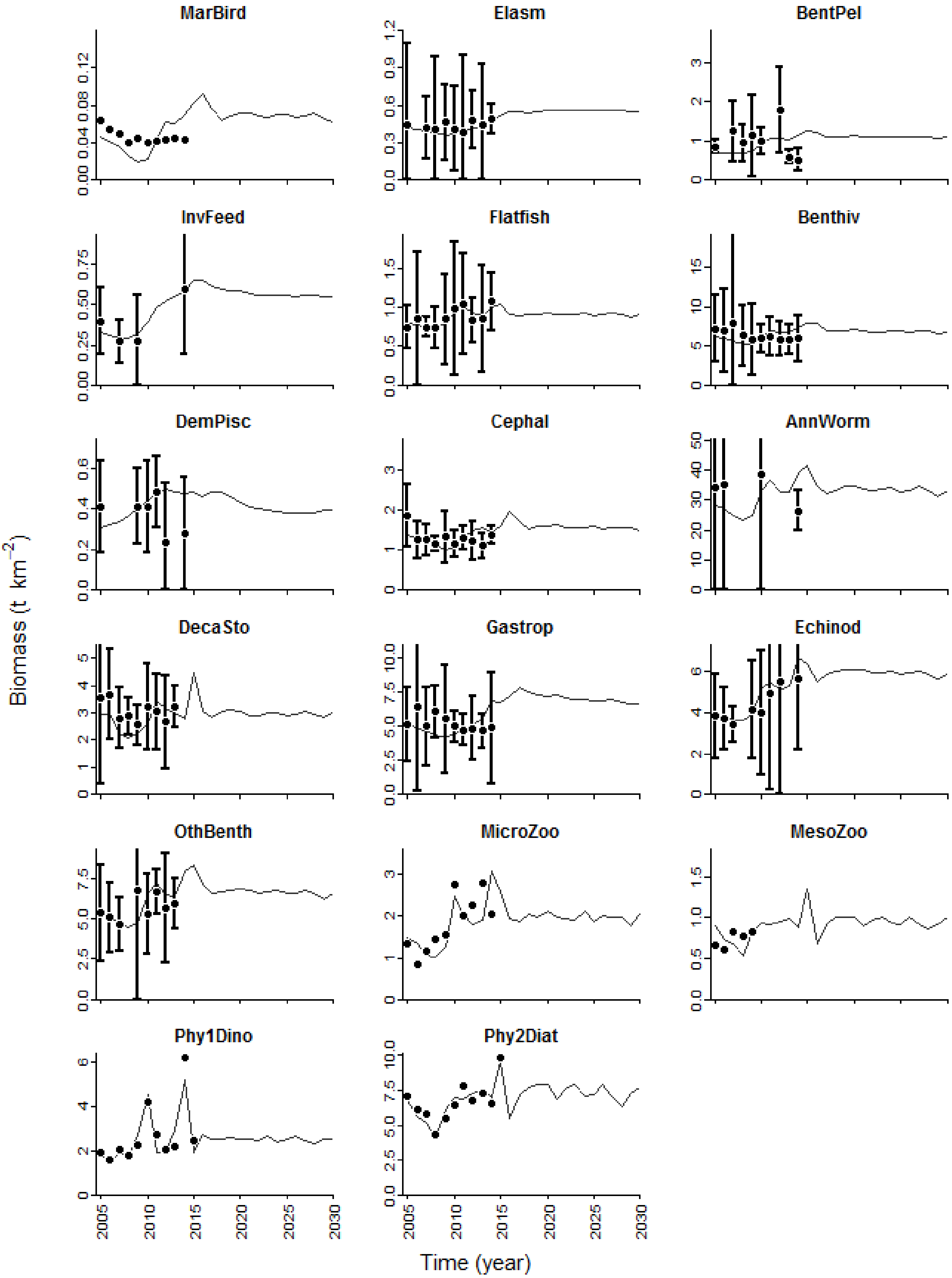

Figure 15. Fitting of refrence biomass simulations. Model predicted values are represented by the line. Relative observed biomasses and their standard deviation are shown as dots and error bars. Abbreviations as in Table 6.

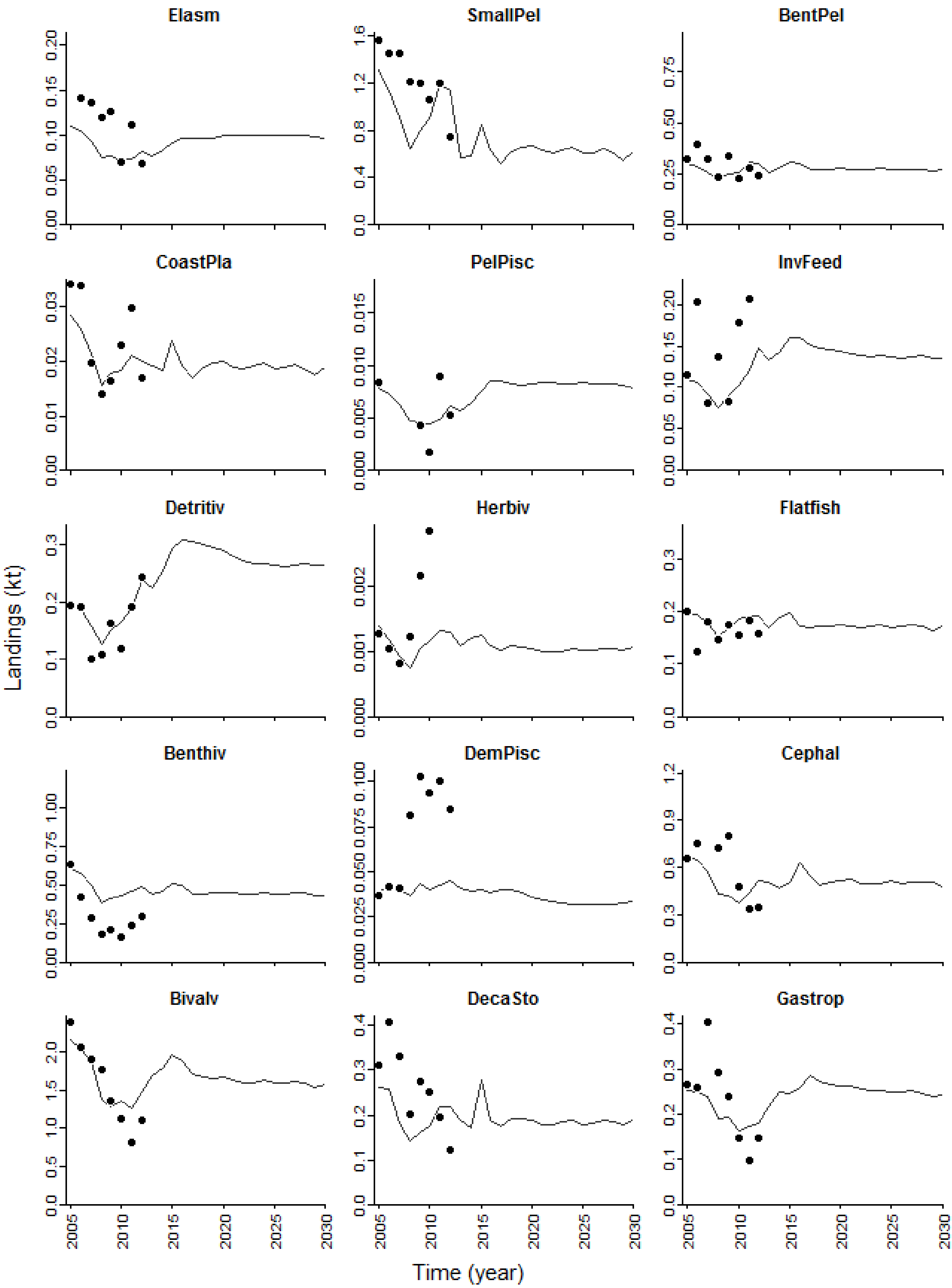

Figure 16. Fitting of refrence landings simulations. Model predicted values are represented by the line. Absolute observed landings are shown as dots. Abbreviations as in Table 6.

5.2. Analysis of Ecosim model’s sensitivity to parameters and forcings

A preliminary analysis on the sensitivity of the time-dynamic model was carried to assess the influence of the initial conditions, the production and consumption basic parameters, as well as different fishing efforts. In the first two cases a Monte Carlo analysis allows for selecting the combinations of main basic parameters and initial conditions (biomasses) that allow to met the mass balance assumption of Ecopath, and permit Ecosim simulation. Initial conditions and basic parameters are tested separately and together: in this way the Monte Carlo analysis can be used to both test the sensitivity of Ecosim’s outputs to Ecopath parameters, and to explore the combinations of parameters that can lower the simulation’s SS values.

In the analysis, a common transformation was used to compare the predicted outputs to the reference simulation (REF), the relative difference from reference simulation (Eq. 24).

$$D_{rel} = \frac{D_{sim_{tstep,fg}} - D_{ref_{tstep,fg}}}{D_{ref_{tstep,fg}}} \qquad \text{Eq. 24}$$

Where *D* (generic data) represent either biomass (*B*) or catch (*C*) data.

A similar approach was used to compare the goodness of fit with the reference model (Eq. 25).

$$SS_{rel} = \frac{SS_i - SS_{ref}}{SS_{ref}} \qquad \text{Eq. 25}$$

Where SS_i and SS_{ref} are respectively the trial and reference goodness of fit measured as sum of squared deviations (SS).

5.3. The Monte Carlo analysis on biomass and main parameters

Given a lower and upper limit around the Ecopath biomass or parameter value, the Monte Carlo routine in the Ecopath with Ecosim software randomly searches a set of values for selected functional groups and parameter types that will give an alternative mass-balanced model. The current Monte Carlo analysis allows modifying the initial biomass (B), the production (P/B) and consumption rates (Q/B), the ecotrophic efficiency (EE), the biomass accumulation (BA) and the diets.

Four different tests were carried out to assess the sensitivity of the reference Ecosim model to variation of: a) the initial biomass (B trials), b) the productivity rate (P trials), c) the consumption rate (Q trials) and d) cumulative effects of the previous by simultaneously modifying all the three parameters (BPQ trials). Changes of initial biomasses and productivity rates are allowed for producer and consumer functional groups, while the consumption rate is allowed only for consumers. For each test 1100 runs were carried and the first 1000 mass-balanced Monte Carlo trials were selected (all EE<1). The parameter selection range was within ±10% differences of the base value (for details see Table 5). The mussel farms functional group was excluded from the analysis since its biomass trend is forced, and this way avoiding its influence on the Ecopath model's mass-balance estimation.

The results were evaluated by comparing i) the range of valid input parameters compared to the reference Ecopath model parameters; ii) the effects on biomass of functional groups at the end of the simulation. These analyses allow an evaluation of whether the initial conditions and parameters used in the reference model are centered or not in the range of values that meet the mass-balance requirements. Moreover, the analysis allows for testing if there are sets of parameters that can reduce the SS values (Eq. 20, 21, 22), and finally it permits for evaluating the dispersion of results as a measure of precision of the model.

In trials with negative relative SS values the parameter distribution was analysed by comparing the improvement pattern with the range of the five trials with lowest SS and the error direction index (EDI, Eq. 26, 27, 28). The EDI decomposes the SS in a portion of errors above and below the observed values and indicates the direction in which the parameter value should move to decrease the error for that functional group and balance between the two errors. EDI is formally expressed as:

$$SS_{upper} = \sum\nolimits_{\{tstep/Dpred_{tstep}<Dobs_{tstep}\}} SS_{tstep} \qquad \text{Eq. 26}$$

$$SS_{lower} = \sum\nolimits_{\{tstep/Dpred_{tstep}>Dobs_{tstep}\}} SS_{tstep} \qquad \text{Eq. 27}$$

$$EDI = \frac{SS_{upper} - SS_{lower}}{SS_{upper} + SS_{lower}} \qquad \text{Eq. 28}$$

Where $D_{pred_{tstep}}$ and $D_{obs_{tstep}}$ are the model's simulation predicted and the observed biomass or catch at a certain time-step.

The influence of the different scenarios of parameter variation on the fit, correlation and standard deviation of each trial was investigated by comparing the biomass predicted in each trial with observed biomass using the Taylor diagrams. Additionally, the susceptibility to parameter variation of each functional group was evaluated by measuring the whole simulation standard deviation of relative biomass values (Eq. 29).

The influence of parameter's variation on the biomass simulation of each functional group was assessed analysing the slope's variation and correlation to parameters values. For that purpouse the value of the slope of an imaginary line connecting the corrected biomass at the beginning of the simulation and the corrected biomass value (B_{corr}) at the end of the simulation (Eq. 30) is used. The corrected biomass (Eq. 30) gives an unbiased proxy to changes of initial biomass. By applying this correction, the dominant effect of changing the initial biomass, and its influence on the whole simulation (changes of the origin of the biomass simulation), is removed. This way, the influence of changes in initial parametters (B, P, Q) can be analysed through trophic interactions affecting biomass simulations, which otherwise would be masked.

$$SD_{whole_{fg}} = \sqrt{\frac{1}{N_{tstep}} \sum\nolimits_{tstep=1}^{N} \left(B_{rel_{tstep,fg}} - \overline{B_{rel_{fg}}}\right)^2} \qquad \text{Eq. 29}$$

$$B_{corr} = \left(\frac{1}{N_{last\ 5\ years}} \cdot \sum\nolimits_{year=2026}^{2030} \frac{B_{sim_{year}}}{B_{ref_{year}}}\right) - \frac{B_{sim_{init}}}{B_{ref_{init}}} \qquad \text{Eq. 30}$$

The corrected biomass (B_{corr}) was calculated by taking mean relative biomass in the last five years of the simulation and subtracting the relative variation of the Ecopath initial biomass ($B_{sim_{init}}/B_{ref_{init}}$). This way a correction for initial biomass change was applied to trials with changes in B and BPQ, while for the trials P and Q there was no need for such correction.

5.3.1. Results of Monte Carlo analysis on biomass and main parameters

The results from the Monte Carlo (MC) trials obtained by changing biomasses (trials B), productivity (trials P) and consumption rates (trials Q) within the ±10% range simultaneously for different functional groups indicate that parameters resulting in valid model have their means centred on the Ecopath base value and a standard deviation smaller than 5% (Figure 17). Exceptions were the Elasmobranchii (Figure 17.a,b,d,e), Benthopelagic fish (Figure 17.c), Benthivorous fish (Figure 17.a,c,d,f), Decapoda & Stomatopoda (Figure 17.a,b,c,e,f), Bacterioplankton (Figure 17.a,b,c,e,f), and Diatoms (Figure 17.a,b) whose means diverged from the Ecopath base value. Moreover, in general, parameters of MC trials with valid Ecopath solutions had a normal-like distribution around the mean value.

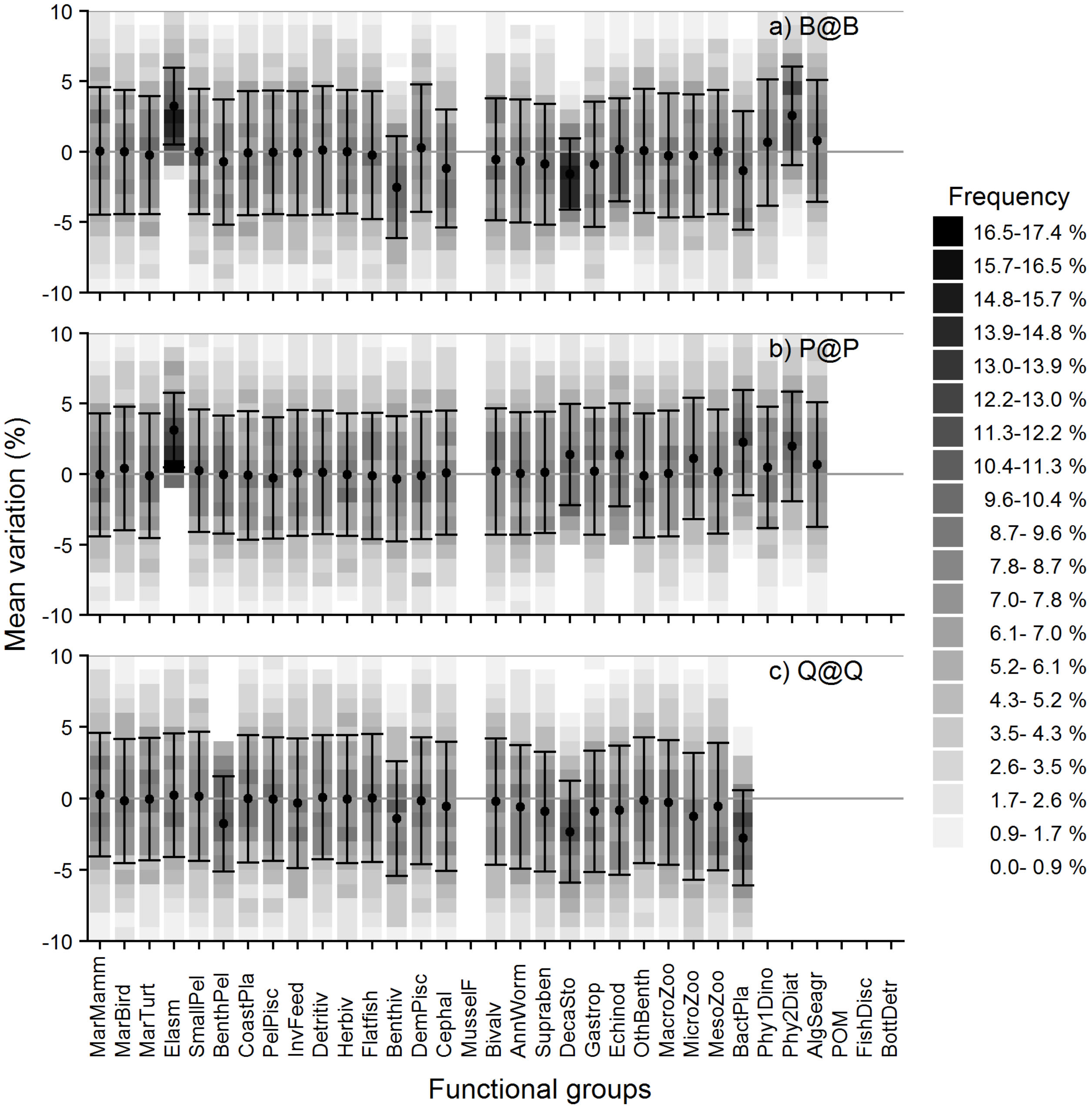

Figure 17. Summary of Monte Carlo parameters. The mean value and the standard deviation are represented by the black points and the error bars. The frequency of the found values in each bin is represented in grey scale. Test trials: a) - only initial biomass (B), b) - only productivity rate parameter (P), c) - only consumption rate parameter (Q) and d ,e ,f) – simultaneously all the parameters (BPQ). Abbreviations as in Table 6.

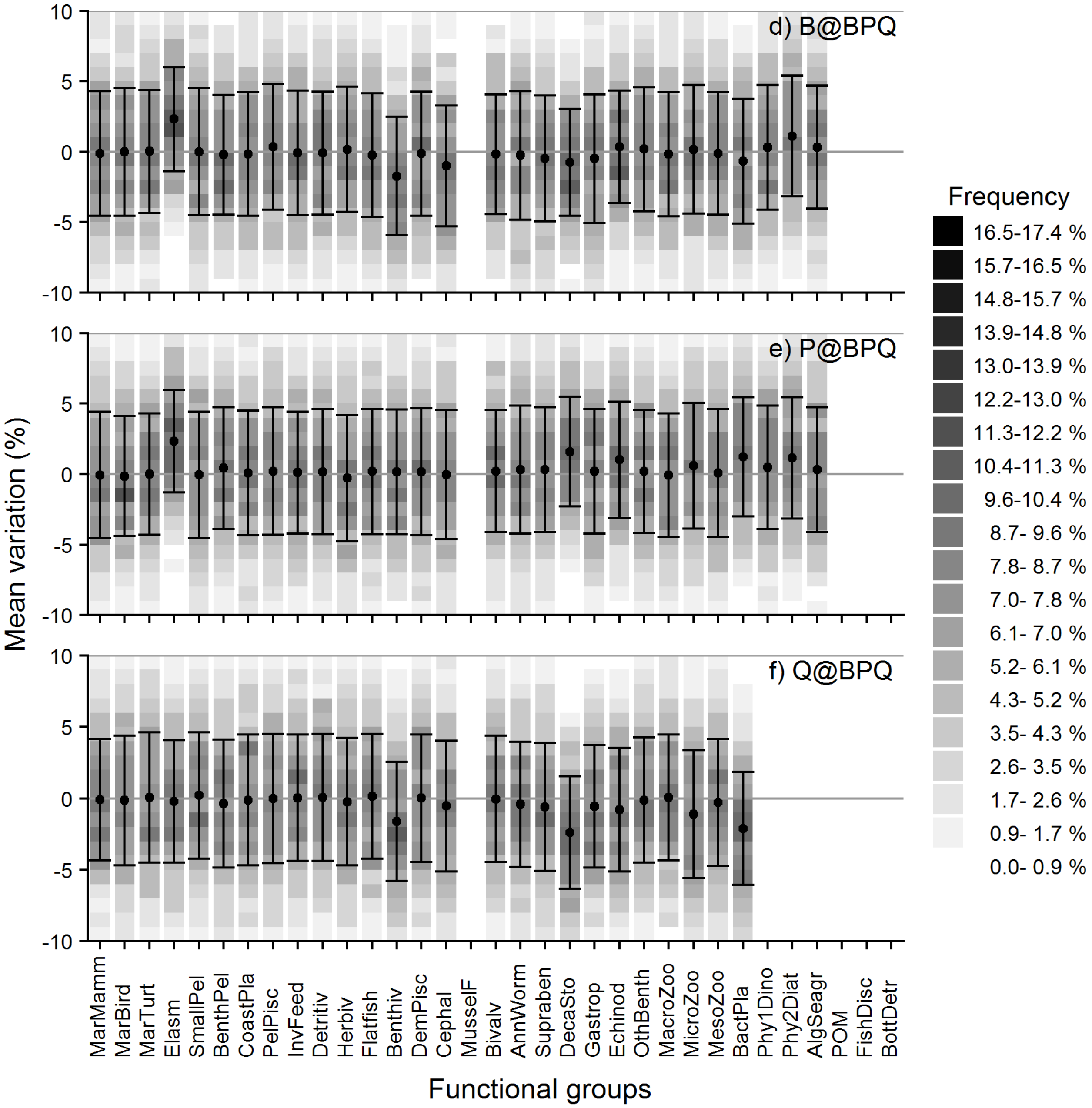

Figure 17. Summary of Monte Carlo parameters. (continuation)

However, the resulting parameters had a different effect on the SS values in different test trials (Figure 18). In the B trials, the SS values had a normal-like distribution balanced around the reference SS value, the range of values was between –4.5% and +5.5% and a moda at +0.5%. P and Q trials had a lognormal-like distribution with higher relative SS values than the reference model. Their ranges were respectively –1.0% - +7.5% and -0.5% - +8.5%, and the modas were both at 0.5%. In the BPQ, trials the variation of SS values indicates an intermediate distribution between the previous three cases, having also the largest range (-4.5% - +24.0%) and the moda at 1.5%. Additionally, the largest number of trials with negative relative SS was observed in B trials (40.7%) and the lowest in the Q trials (6.7%).

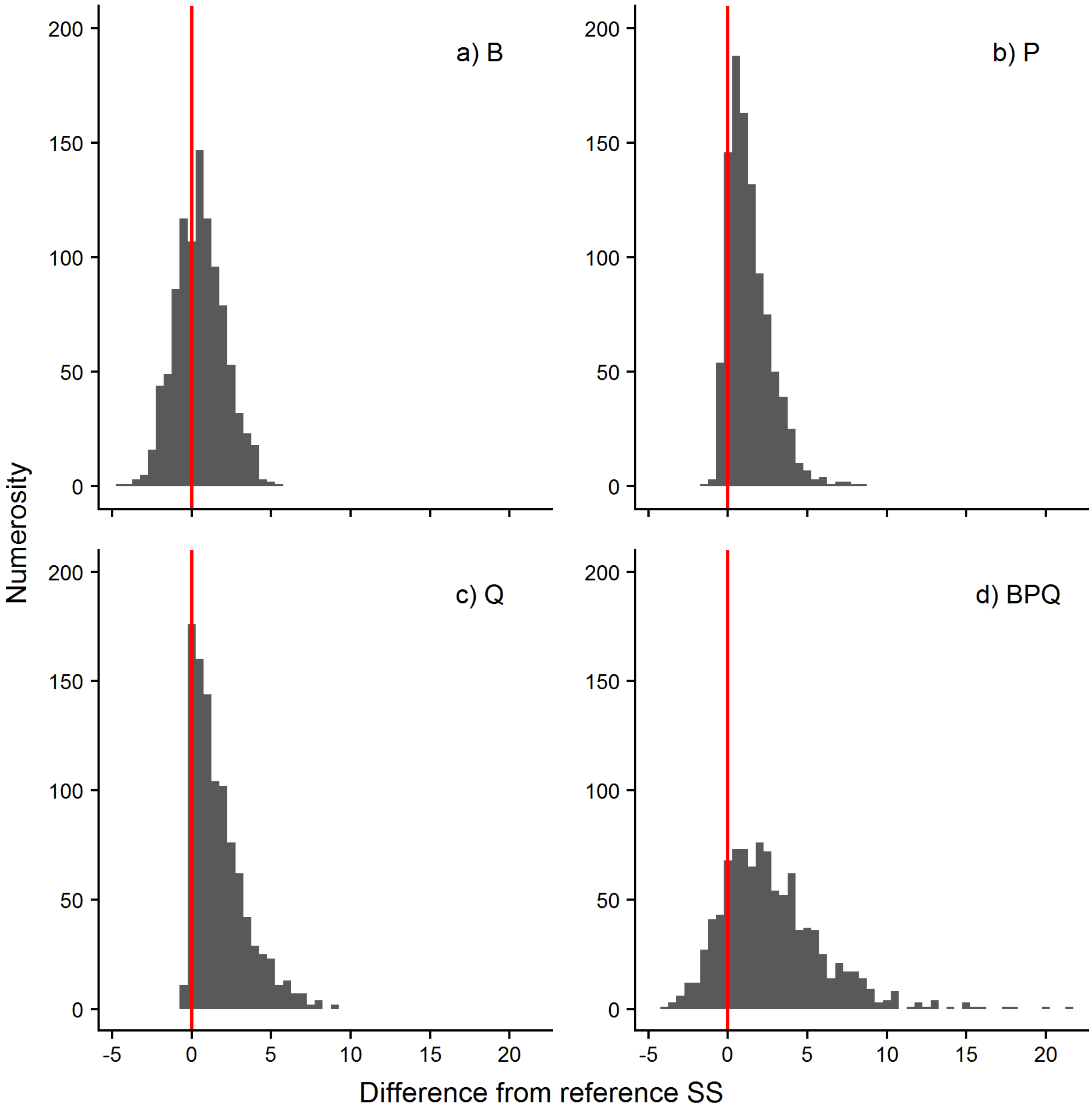

Figure 18. Distribution of SS values from Monte Carlo trials. The reference SS value is represented by the red line and the bin size is 0.5%.

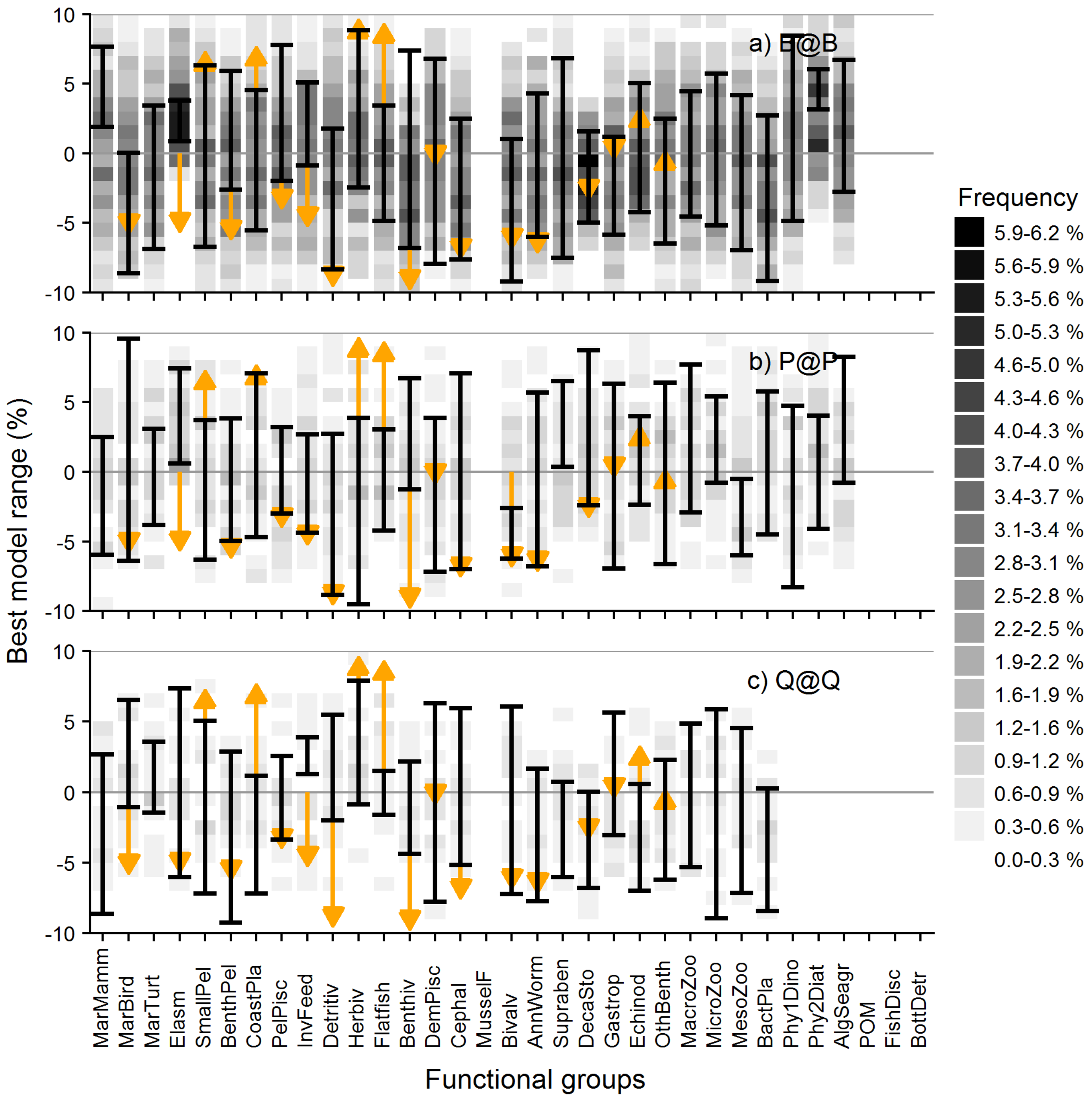

Figure 19. Summary of the parameter ranges and distribution of trials with negative relative SS. The ranges of the five trials with lowest SS are represented by the line range bars. The error direction index is represented by the yellow arrows on a separate scale [-1,1]. The frequency of the trials in each bin is represented in grey scale. Abbreviations as in Table 6.

The analysis of trials with negative relative SS (i.e. trilas with an improved fit in comparison to the reference model) indicates different patterns can be observed between the four MC trials (Figure 19). A pattern where the highest frequencies of trials with negative relative SS are in parameter ranges offset from the Ecopath base value was the clearest in the B trials, less pronounced in P and Q trials, and the least in BPQ trials. Additionally, the parameter solutions of the five trials with the lowest SS show a wide range among all test trials. In exceptions like Bacterioplankton (Figure 19.a), Bivalvia (Figure 19.b) and Benthivorous fish (Figure 19.d) a narrower range of parameter solutions was identified. Some of parameter solutions of the five trials with the lowest SS had solutions in the direction of the major error (EDI), but often the range of these five trials did not follow that same direction. Some possible solutions that reduce the overall SS values were also found in opposite directions of the EDI and at extreme ends of the search ranges.

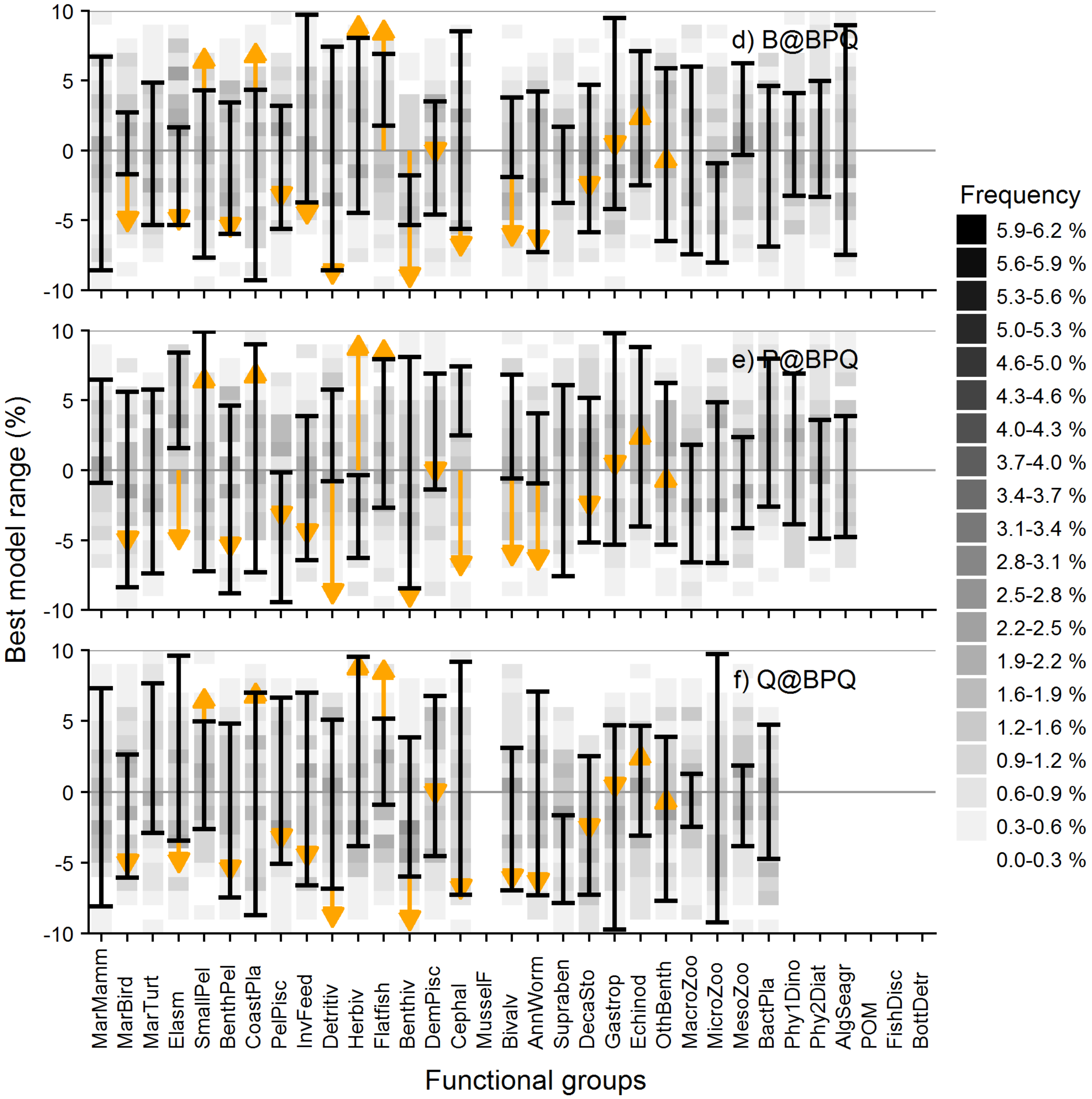

Figure 19. Summary of the parameter ranges and distribution of trials with negative relative SS. (continuation)

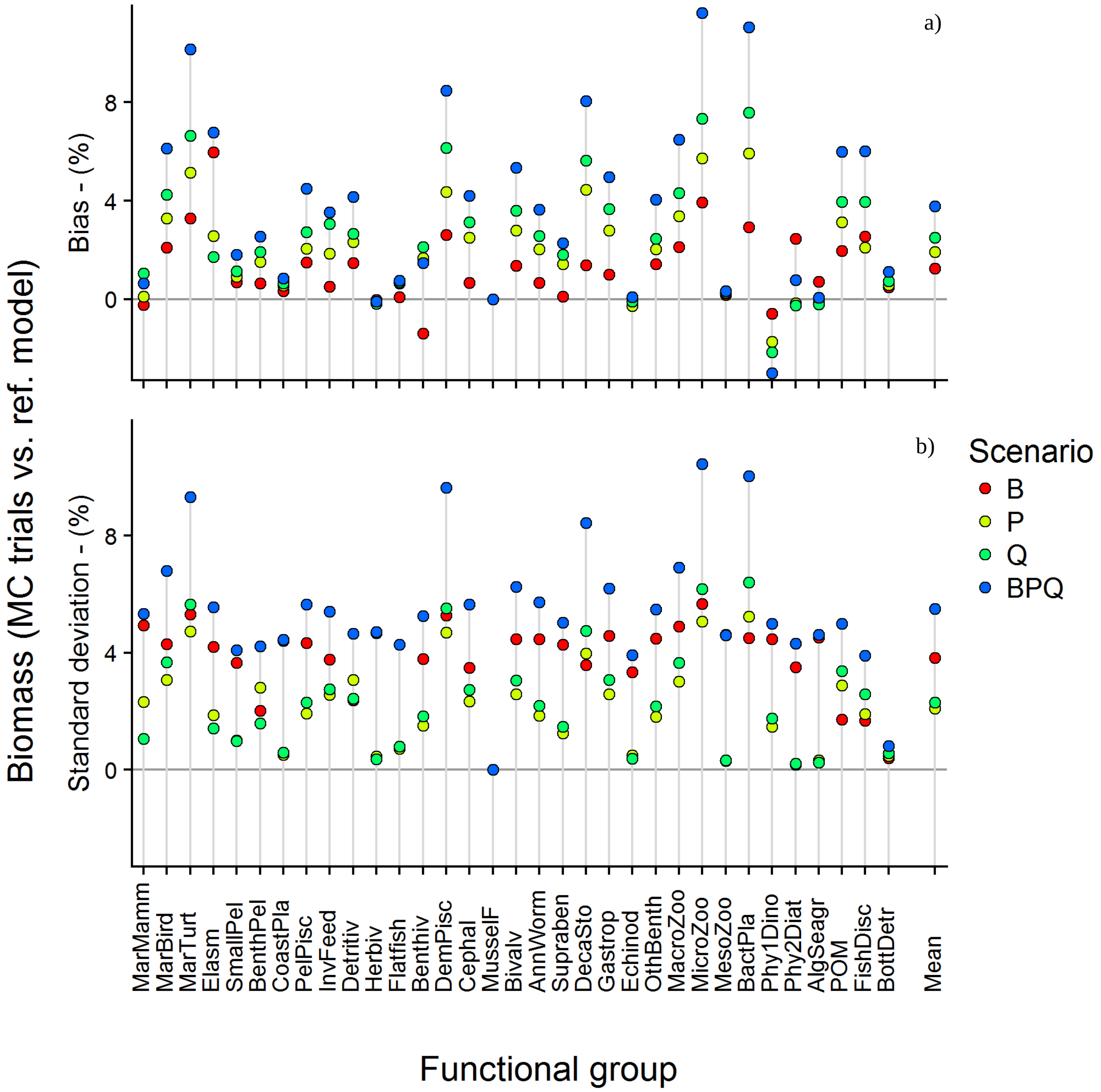

Figure 20. Mean biomass difference from the reference simulation (bias) and variation for each functional group at the end of simulations in the four tests. Results of mussel farm should be ignored. Abbreviations as in Table 6.

The MC analysis allows for estimating bias and dispersion of MC trials with respect to reference simulation for each MC trial (B,P,Q and BPQ). The bias indicates the trials' mean shift from the reference simulation value, while the mean dispersion around the reference value is indicated by the standard deviation. Both have a similar trend across the functional groups, but the bias indicates the mean direction of the shift. These results showed that parameter also had different influences on the variation of functional group biomasses at the end of the simulations (Figure 20.). Generally, the mean of all functional groups indicates the BPQ trials had the biggest influence both on biomass bias from the reference Ecosim model (3.6%, Figure 20.a) and biomass variation (5.5%, Figure 20.b). Q and P trials had the second largest influence on bias (2.4% and 1.9%, respectively), and B trials had the least (1.2%). Instead, B trials had the second largest influence on biomass variation (3.8%), while Q and P trials had the least influence (respectively 2.3% and 1.9%). All functional groups had the influence of P, Q and BPQ trials pointing to higher bias values than B trials, exceptions were the three primary producers, where the P, Q and BPQ trials were pointing to lower values than B trials. Additionally, in Dinoflagellates all trials were pointing to decreasing biomasses. The analysis highlighted that the responses of Marine turtles, Demersal piscivorous fish, Decapoda & Stomatopoda, Microzooplankton and Bacterioplankton had the largest variations in all four test trials. On the other hand, Marine mammals, Small pelagic fish, Coastal planktivorous fish, Herbivorous fish, Echinodermata, Algae and seagrass and Bottom detritus had the lowest biases. The mentioned functional groups had also the lowest values of biomass variation in P and Q trials, while only the Bottom detritus was the least influenced in all the trials.

Moreover, the same biomass variation at the end of the simulations (2026-2030, Figure 20.b) was also observed during the hindcast (2005-2015, Taylor diagrams, Figure 21). The lowest variability was observed in trials Q and P, and the highest in BPQ trials. It is possible to notice that the predicted values of variation of each functional group follow a radial pattern and are larger than the tangential variation of the correlations.

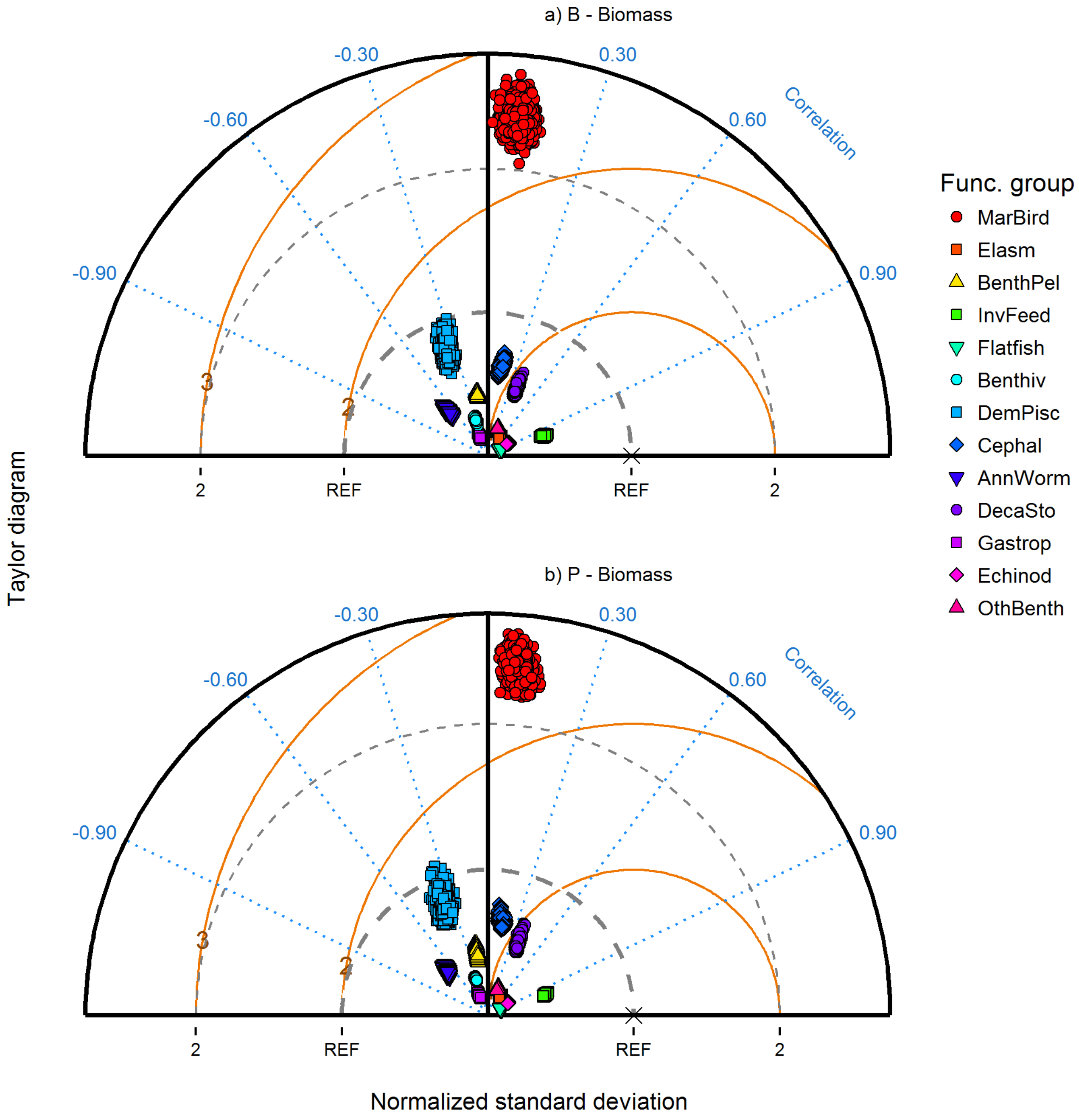

Figure 21. Taylor diagrams of functional groups' biomass variation at the end of trials' simulations. The pointer's distance from the origin indicates the standard deviation of predicted data (grey lines); the observed data standard deviation is represented by the thick grey line as reference (REF); the distance from the observed data (REF with a cross) indicates the unbiased relative mean standard deviation (uRMSD) of predicted data from observed data (orange lines); the angle formed between the pointer and the origin indicates the correlation od predicted data with observed data (blue lines). Abbreviations as in Table 6.

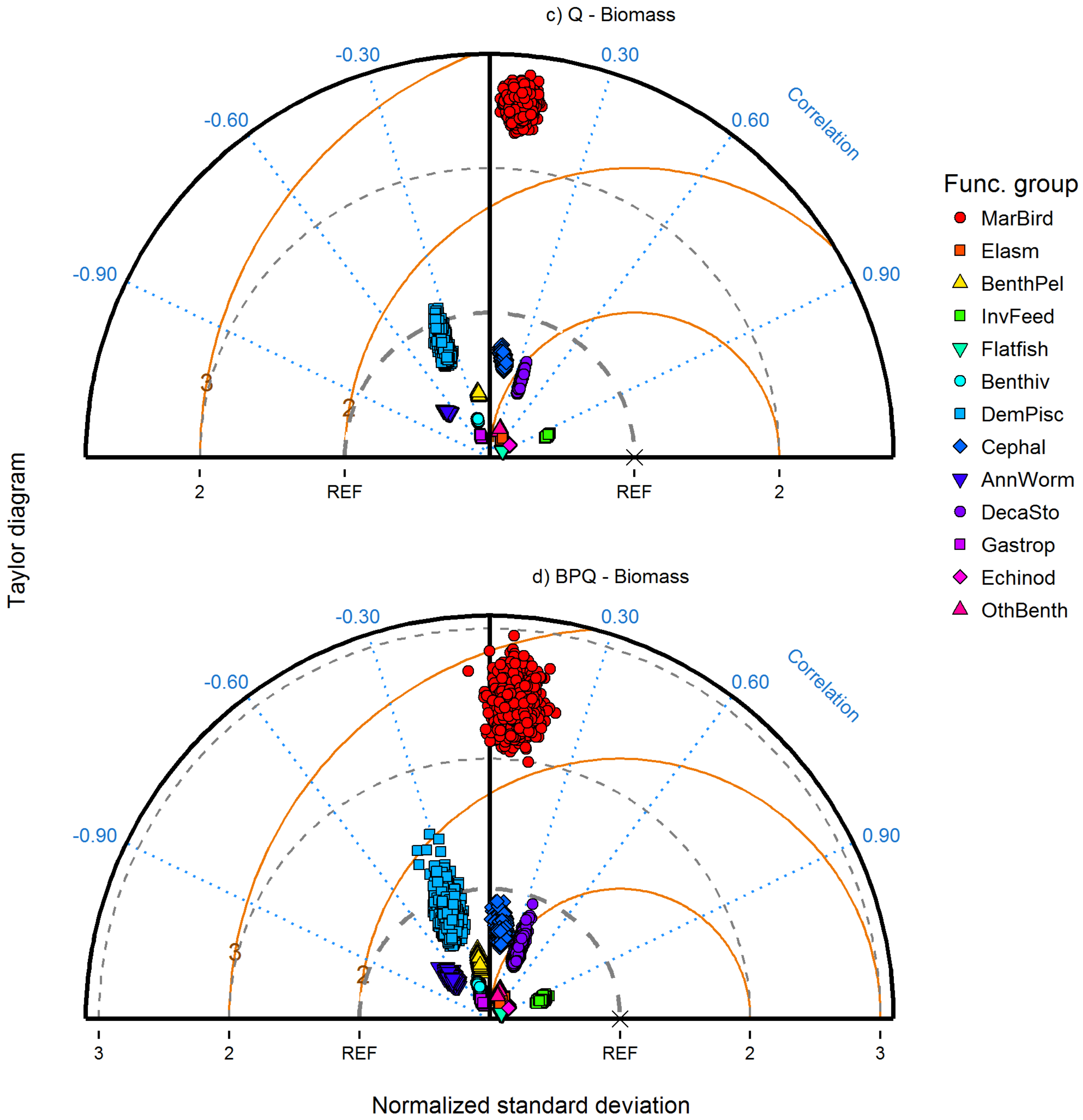

Figure 21. Taylor diagrams of functional groups' biomass variation at the end of trials' simulations. The pointer's distance from the origin indicates the standard deviation of predicted data (grey lines); the observed data standard deviation is represented by the thick grey line as reference (REF); the distance from the observed data (REF with a cross) indicates the unbiased relative mean standard deviation (uRMSD) of predicted data from observed data (orange lines); the angle formed between the pointer and the origin indicates the correlation od predicted data with observed data (blue lines). Abbreviations as in Table 6.

Moreover, the functional groups showed different behaviour to variation of the parameters in the four tests (Figure 22). Analysis of the whole simulation mean biomass variation was carried to asses how much a functional group's biomass oscilates during the simulation (Figure 22.a). Results indicate the largest variation was observed in Marine birds (up to 5.1%), Marine turtles (up to 3.0%) and Demersal piscivorous fish (up to 2.9%) (Figure 22.a). The BPQ trials consistently showed the largest variation among the four tests (mean variation 1.40%), while the others had a mixed behaviour and their means were overlapping (respectively the mean variations: B: 0.71%, P 0.82%, Q 0.76%).

The analysis of the slope (corrected biomass at the end of the simulation; Eq. 30) deviations of the simulation, indicates functional groups had heterogeneous susceptibility and size to parameter variation in the four tests (Figure 22.b). Major deviations were observed in Marine turtles, Demersal piscivorous fish, Decapoda & Stomatopoda, Microzooplankton and Bacterioplankton, while on the other hand, Herbivorous fish, Flatfish, Echinodemata, Meso-zooplankton, Diatoms, Algae and seagrasses and Bottom detritus had low mean deviation. Among the tests, the BPQ trials had the largest mean deviations (3.8%) followed by the trials Q, P and B (respectively 2.3%, 1.9%, 1.6%).

Analysis of functional groups' correlation between the deviations of their simulation's slope (corrected biomass at the end of the simulation; Eq. 30) and the MC parameters (Figure 17) is illustrated in Figure 22.c. Functional groups with highest correlations were the Small pelagic fish, Benthopelagic fish, Detritivorous fish, Herbivorous fish, Cephalopoda, Echinodermata, Mesozooplankton, Bacterioplankton and Algae and seagrasses. Except once for the Q trials, correlations were mainly observed in B and P trials. Q and BPQ trials had low to no correlations. Mean r^2 values indicate only trials B and P had some influence on the deviations of the simulation's slope (respectively r^2=0.207 and r^2=0.174), while the other four combinations of parameter variation in trials Q and BPQ had low correlation (r^2=0.053 Q@Q, r^2=0.084 B@BPQ, r^2=0.055 P@BPQ, r^2=0.013 Q@BPQ).

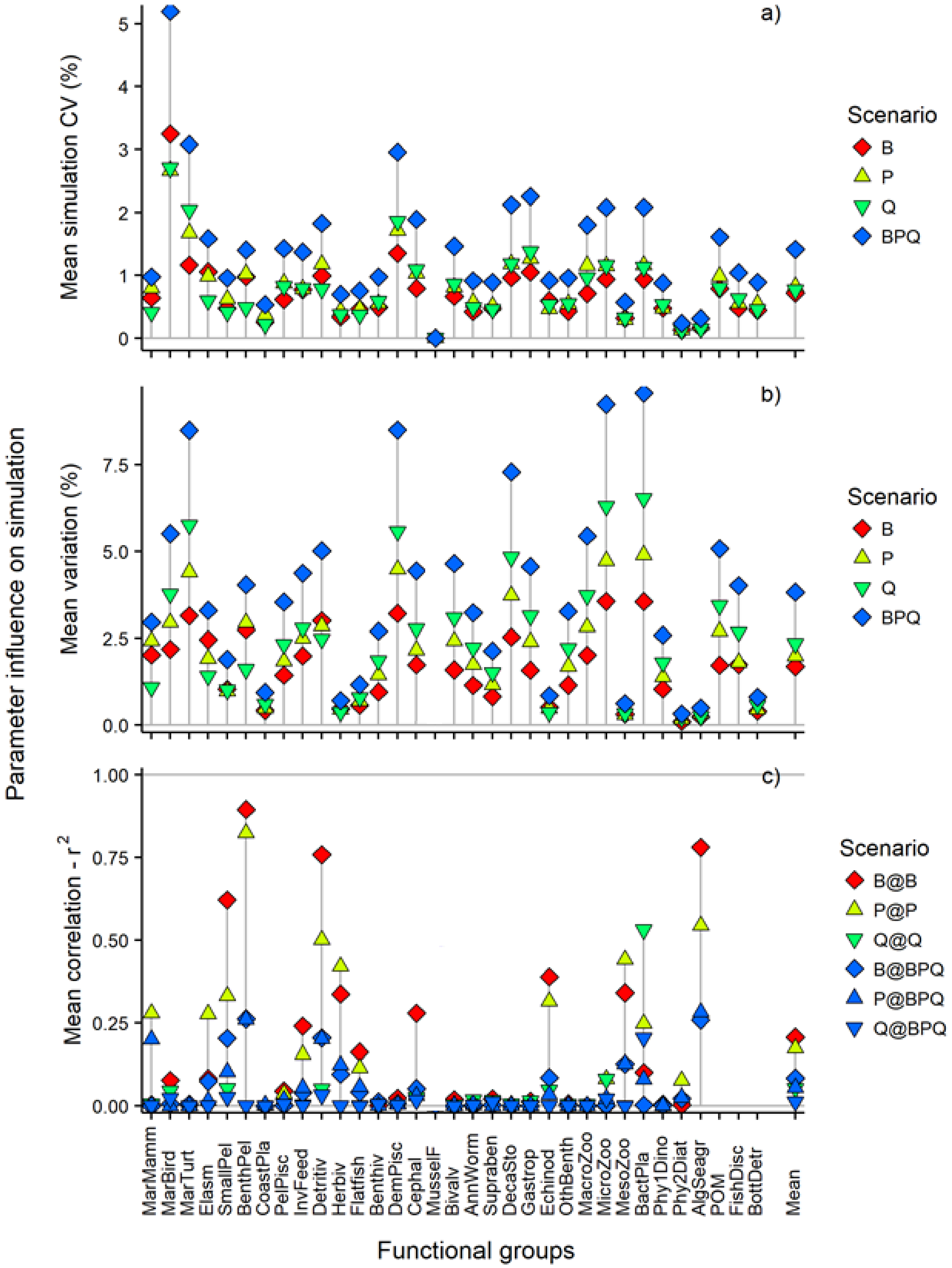

Figure 22. Summary of the influences the parameter variations have on functional groups' simulations. Panels: a) standard deviation of whole simulation's relative biomasses, b) mean of standard deviations of corrected biomasses at the end of simulation and c) means of correlations between corrected biomasses at the end of the simulation and the Monte Carlo parameters. The dots represent the mean value for each test trials. Abbreviations as in Table 6.

5.4. Impact of different fishing effort measures on model results

Fishing effort indicates the level of pressure the fishing fleet exerts on the exploited resources and the ecosystem. Measuring effort is not straightforward since it is a conventional variable. In fact, it depends not only by the number of fishing vessels, their capacity, the hours spent fishing, but includes also technical characteristics of the gear (i.e., with larger gears the effort is larger in the same unit of time) and fishing operations (e.g., speed of the vessel). Moreover, different resources are caught by different fishing tools; therefore, different indicators of fishing capacity may be more suitable to express the variation of a fishing effort on certain species. Although conventionally power of vessel (kW) by number of hours spent fishing (days at sea or an equivalent indicator) is a common measure of effort for trawlers (FAO, 2016), this is not the case for other fishing devices. Thus, it is inderterminate, which is the most appropriate measure for the quantification of effort, and using one measure might result in model misspecification.

The following analysis was thus conducted to assesses how different fishing effort measures influence the biomass of functional groups in the Ecosim model simulations. For that purpose, fishing effort levels per fishing gear were calculated using three different measures of fishing capacity as proxies: the number of fishing vessels, the cubic LOA (an indicator of the volume of the fishing vessel) and the engine power (kW). Fishing effort of four fishing gears of both Italy's and Slovenia's fleets were modified for this test: bottom otter trawl (OTB), midwater pair pelagic trawl (PTM), purse seine (PS) and small-scale fisheries (SSF). Rapido trawl's fishing capacity did not change over time and dredges operate under a regulated quota system so the vessel's characteristics do not influence the effective fishing effort, therefore these two fleets were excluded from the analysis.

The fishing effort was calculated using the Equation 31, being the fishing capacity the only time series per gear that changed. Fishing effort time series were expressed relative to their first year's initial value in 2005.

$$E_{tstep,fl} = fishing\ capacity_{tstep,fl} * \ days\ at\ sea_{tstep,fl} * \ seasonality_{tstep,fl} \qquad \text{Eq. 31}$$

The resulting test scenarios consisted in 81 (3^4) trials. The reference simulation had fishing effort based on cubic LOA for OTB and PTM, and number of vessels for PS and SSF, which was also one of the possible combination in those 81 trials. The effects of fishing effort changes were investigated by comparing trials SS values and the biomass variation (Eq. 24) at the end of the trial simulations with the reference simulation.

5.4.1. Effects of application of different fishing effort measures on model results

The largest variations of fishing effort were observed for Slovenia’s bottom otter trawl fleet and Italy’s purse seine and small scale fishery fleets, while the fishing effort Slovenia’s small scale fishery fleet had the smallest variation on changes of the fishing capacity indicator (Table 11).

The effects of different fishing effort measures had little influence on the simulations’ relative SS values, which changed less than 0.3% (Figure 23).

Table 11. Variation of the fishing effort at the end of the simulation compared to the reference simulation. Fishing capacity indicators are the number of vessels (Num), cubic length-over-all (LOA^3) and engine power (kW). Fishing gear codes: bottom otter trawl (OTB); midwater pelagic trawl (PTM); purse seine (PS); small-scale fisheries (SSF).

Indicator	SLO-OTB	SLO-PTM	SLO-PS	SLO-SSF	FVG-OTB	FVG-PTM	FVG-PS	FVG-SSF
Num	13.8%	-	**0.0%**	**0.0%**	7.5%	7.9%	**0.0%**	**0.0%**
LOA^3	**0.0%**	-	6.6%	-0.1%	**0.0%**	**0.0%**	-10.0%	6.8%
kW	8.9%	-	5.1%	-3.1%	5.5%	5.0%	-0.8%	11.8%
St.Dev.	7.0%	-	3.5%	1.8%	3.9%	4.0%	5.6%	5.9%

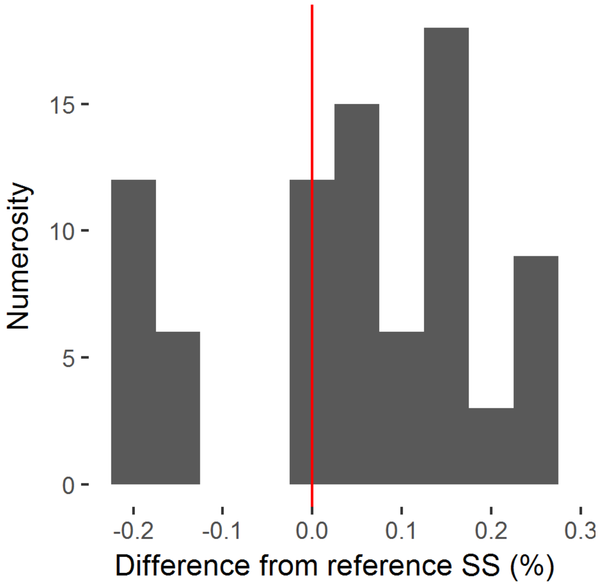

Figure 23. Distribution of SS values in fishing effort trials. The reference SS value is represented by the red line and the bin size is 0.05%.

Different fishing effort measures had major influences on five functional groups: Marine mammals (1.48%), Elasmobranchii (1.98%), Benthopelagic fish (1.39%), Invertebrate feeding fish (3.15%) and Detritivorous fish (2.93%) (Figure 24). Other functional groups had a variation smaller than 1% of the reference biomass. This result gives insights on the groups most sensitive to uncertainty in fishing effort and Figure 24 allows the quantification of the precision of obtained biomasses.

Figure 25 represents the trend of relative biomasses (comparison of simulated biomasses relative to the reference simulation) of functional groups most affected by changes of fishing effort. In all four functional groups the simulations branch in three major trends, corresponding to three fishing effort trends of the most influential fishing gear (small-scale fishery, SSF, for Elasmobranchii, Invertebrate feeding fish and Detritivorous fish; otter trawling, OTB, for Benthopelagic fish). Similarly, each major branching has additional three intermediate branching indicating a second fishing gear's influence (OTB for both Elasmobranchii and Invertebrate feeding fish), although much smaller than the previous one. A bundle of 9 close related simulations indicates the negligible influence of the remaining two fishing gears. In all four functional groups indicated above (Figure 25) the variation of the fishing effort had a negative impact on the biomass simulations, where the largest decrease was observed for Invertebrate feeding fish, more than -8.0%.

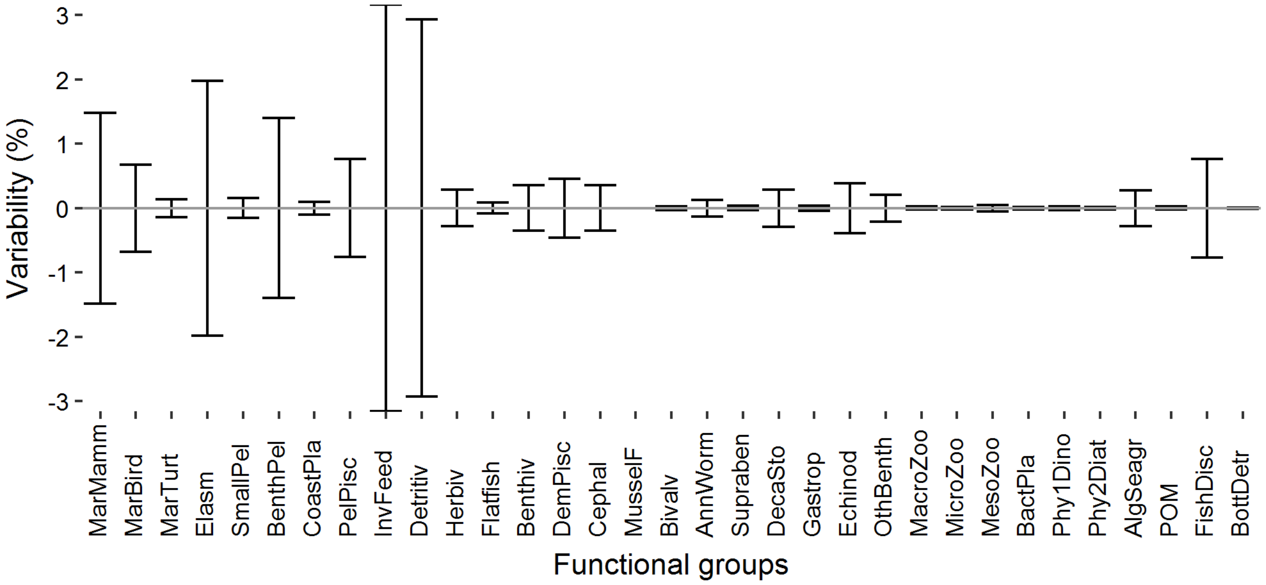

Figure 24. Summary of functional groups' relative biomass variability in all fishing effort changes trials.

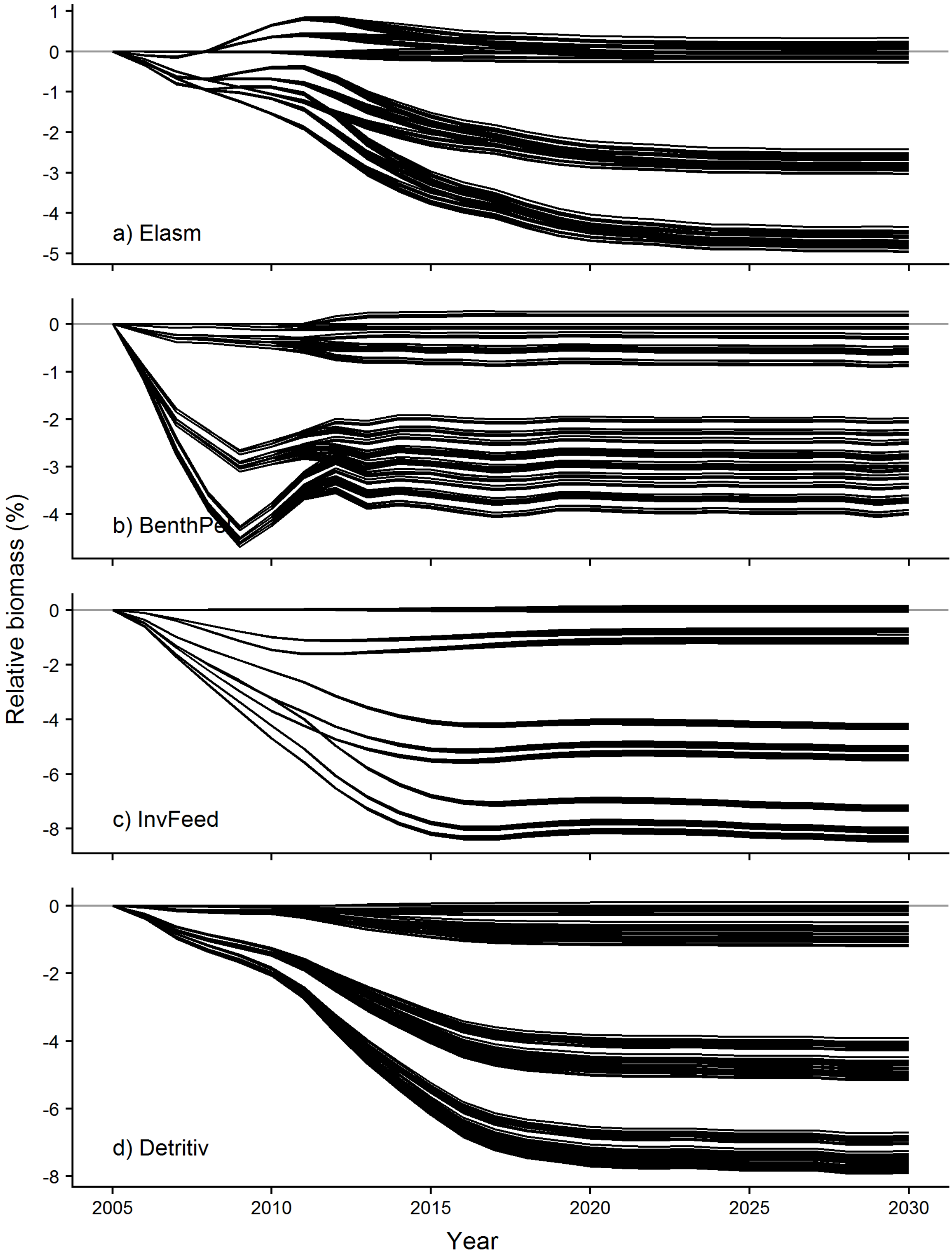

Figure 25. Functional groups' biomass simulations influenced by the variation of fishing effort as a result of using different measures of fishing capacity. Each functional group has 81 simulations, represented by individual black lines (may be overlapped). Labels: Elasm – Elasmobranchii, BenthPel – Benthopelagic fish, InvFeed – Invertebrate feeding fish, Detritiv – Detritivorous fish.

5.5. Discussion

The application of Monte Carlo analysis allowed to assess the influence of uncertainty in forcings (fishing effort measure), main basic parameters (P/B and Q/B) and initial conditions (initial biomasses) on the time-dynamic Ecosim model of the NEAS. The results also gave a general indication of model's precision as influenced by the difference factors and distinguished by functional group.

The Monte Carlo analysis indicates that the valid input parameters were generally distributed in a normal-like pattern around the initial Ecopath value. When a functional group's ecotrophic efficacy is close to 1, however, a series of parameter values are not meeting the mass-balance conditions resulting in a skewed distribution of valid inputs (Figure 17). This case illustrates the variation in the parameters of top predator, which is loosely connected to other functional groups and is represented in our model by the Elasmobranchii, whose ecotrophic efficiency, close to 1. The variation of the initial biomass and the production rate and allows them only to increase to met the criteria EE<1 for their preys. In such situation the Elasmobranchii are able to increase almost freely from their initial biomass because they are neither bound by higher predator control or the predation mortality exerted on other functional groups.

Another case is represented when the variation in predator parameters is limited by its prey. An example is the interaction between two predators, Benthivorous fish and Cephalopoda, and a prey, Decapoda & Stomatopoda. In our model the Decapoda & Stomatopoda are bound by their ecotrophic efficiency close to 1. Therefore, in order to maintain a mass-balance of the prey the increase of the biomass or consumption rates of the two predators are limited, and therefore more solutions are found when decreasing these parameters. In a third case the trophic interaction can be even caused by a detritus group, as in the example of Bottom detritus. In this case the ecotrophic efficiency of Bottom detritus is almost 1, and therefore strongly influencing the output parameters of all three primary producers and its consumer, the Bacterioplankton. Any decrease in the biomass or production rates of primary, primarily the Diatoms, or an increase in the biomass or production rates of Bacterioplankton would void the mass-balance rule. Additionally, the previously mentioned interactions were attenuated when all three parameters were changed simultaneously.

The sum of squared deviations (SS) of valid models of MC trials was not always symmetrically distributed around the reference SS value. In particular, the B trials had a normal-like distribution centred on the reference SS value, while MC trials for P and Q displayed a lognormal-like distribution of SS values. MC trials for BPQ, where effects of simultaneous changes in B, P and Q were integrated, resulted in an even more skewed distribution of SS values for valid models.

The five MC trials with the lowest SS showed parameters close to the reference values, supporting the reliability of parameter setting of Ecopath (reference input parameters). Moreover, the large parameter ranges of five trials with the lowest SS indicate there are no specific parameter regions for such solutions and also might not even be in the direction indicated by EDI.

The biomass bias of functional groups in the trials compared to the reference simulation exhibited a complex non-linear pattern with positive biomass biases, exception being the primary producers which displayed the opposite pattern. A contributing factor may be the fact that Ecopath mass balance is solved using a top-down approach, as opposed to bottom-up techniques as the inverse modelling (Steele, 2009). However, the combination of the properties of the mass-balanced approach, the model constraints for the initial values and the vulnerability values (set at opportune different values for each trophic interaction) are all implicated in these results. Furthermore, the model is designed as a closed system, i.e., with minimal dynamic flows across its boundaries because net migration and outflows were set to zero. Also, the detritus group was modelled with no export, and majority of it is recycled back to the food web, resulting in ecotrophic efficiency close to 1. As a consequence, the input parameters selected in the Monte Carlo were necessarily bounded to allow only an increase of Bottom detritus. The key players in this balance are the primary producers and the Bacterioplankton, as the largest consumer of detritus (950 t km^{-1} y^{-1}) and the holder of the largest consumption in the whole food web. In fact, in all the trials the mean parameter outputs indicate either an increase of the biomasses or production rates of primary producers, increasing the flow to detritus. Similar effects leading to increasing the detritus biomass were achieved by Bacterioplankton's mean input parameters, and the observed correlation of slopes in Figure 22.c might support that. Both the reduction of initial biomasses in B trials and the reduction of consumption rates in Q trials were leading to a decreased consumption of the detritus and its recycle to the food web that caused an initial accumulation in the ecosystem. In case when the Bacterioplankton production rate increased more bacteria were produced and as a consequence larger quantities of dead organisms were flowing to detritus. All these contributing effects of Bacterioplankton and primary produces led to an initial accumulation of detritus in the food web, triggering a delayed effect where its increased biomass could sustain larger biomasses of its consumers. This way the food web reached a new equilibrium and recycled back the detritus. The scale of functional groups' responses to this newfound equilibrium might have been determined by the vulnerabilities parameterisation. Evidence supporting the biomass has been transferred to higher trophic levels is represented by the increase of Fishery discards, similar in size to the POM. Results also suggest that energy has been transferred more through the benthic macro-fauna than through the pelagic loop. However, the causes that lead to this bias should be further investigated.

The variances in functional groups biomasses allowed to effectively evaluate the sensitivity of functional groups' responses to variation of the input parameters, and this way the analysis provides an estimate for the precision of simulations. Changes of initial biomasses had more impact on the variances of the functional groups biomasses than the changes in production or consumption rates. Additionally, the effects of changing of all three parameters simultaneously combined the influences of each single parameter, had the largest variances of functional groups biomasses, but were smaller than the sum of the single variances. Therefore, uncertainty of parameters is not additive and it is resulting in effects of different magnitudes in different functional groups. The observed sensitivities are also evident in the Taylor diagrams during the hindcast, where the sensitivity responses to variation of output parameters are particularly clear in terms of variance of predicted data and variance correlation.

Although the variation of initial parameters may be consistent, their effect on the variability of trophic responses was limited (Figure 22.a). The relative biomass variance of each trial's simulation was low, indicating reduced trophic responses and more likely shifting the simulation to higher or lower biomass levels. In fact, this is supported by the analysis of the variances of the trials' slopes, where the effects of the shift of initial biomasses were corrected (Figure 22.b). Results indicate the variation of initial biomasses has the lowest influence on the size of the slope, producing variances much lower than those observed without the mentioned correction (Figure 20.b). Therefore, it could be assumed that the variation of initial biomass shifts the predicted biomass level, but has low influence on the slope (Figure 22.c). The inclination of the slope in most of the cases was uncorrelated to the variation of initial parameters, in particular for Q and BPQ trials. Correlations in B and P trials are found only in functional groups which are less constrained by trophic interactions with other functional groups.

On the other hand, the variation of the fishing effort allowed estimating the sensitivity of different fishing efforts on the food web. Except for Slovenia's small-scale fishery and Italy's purse seine fishery, other fishing effort measures were higher than reference simulation's one. However, fishing effort changes had limited effects on the food web because most of functional groups had low fishing mortality. Fish groups had high fishing mortalities (Table 5) and sustained the major effects. The fishing effort of small-scale fishery influenced Elasmobranchii, Invertebrate feeding fish and Detritivorous fish, while the bottom otter trawl fishery influenced the Benthopelagic fish. The observed variance of functional groups biomasses was lower than the variance of the fishing effort, mainly because it was scaled proportionally to fishing mortality's contribution to functional group's total mortality. Additionally, non-fished groups, like the Marine mammals, were also influenced by changes of fishing effort through trophic interaction with their prey, the Benthopelagic fish. The analysis indicates only the top predators and the fish groups are influenced by the selected fishing gears, while the benthic macrofauna is less influenced. Small variations of relative SS values provide evidence of fishing efforts variations’ low influence on the food web.

6. Ensemble of models as a strategy to better represent observations

The present chapter has the objective to develop a method to better represent the intrinsic trophic responses of the food web and to evaluate the proposed solutions for further operational use by analysing the precision and accuracy in representing field data.

Typically, modelling involves selecting a best performing technique from a range of alternatives, however, independent evaluations of models may be unable to justify the pre-eminence of any single one (Araújo and New, 2006). The concept of ensemble has been introduced into statistical mechanics by J. Willard Gibbs in 1878, consisting in a large number of copies of a system considered all at once, each representing a possible state of that system (Araújo and New, 2006). It may be more narrowly defined as multiple simulations where the initial conditions, model classes, parameters and boundary conditions change. Instead of picking the best model from the ensemble it could be more promising to explore the resulting range of projections.

Some of the uses of ensembles of different models in fisheries science were estimating stock status (Rosenberg et al., 2017), evaluating fisheries scenarios (Smith et al., 2015) and modelling species distribution (Araújo and New, 2006; Guo et al., 2015). Ecosim models have been used as part of model ensembles for reconstruction of past variation and future projections, especially when estimating long-term climatic effects (Meier et al., 2012; Gårdmark et al., 2013). However, ensembles of Ecosim models have not been reported yet. Smith et al. (2015) suggest the importance of studying the impact on predators of shifting between alternative food sources, and highlight care should be taken to understand the trophic assumptions of the models. Therefore, here the performance of ensemble of Ecosim models is proposed and tested based on variation of trophic responses.

The proposed NEAS ensemble of models is based on 21 Ecosim simulations, obtained by applying several fitting procedures of trophic interactions. All simulations share the same primary production and fishing effort forcing functions; the vulnerabilities are the only parameter type to be estimated.

The two options for fitting the vulnerability parameters in Ecosim were used to explore model behaviour: a) the selection of vulnerability values and b) the selection of the observation data for fitting. The former option has two possible methods of selecting the vulnerabilities, either i) in the predator-only strategy (P/R) the predator exhibits the same vulnerability for all its prey; or ii) the predator-prey strategy (P/P), which treats individually each prey-predator interaction. Further, the fitting procedure may either i) use all the observation data or ii) optimise specific functional groups by selecting their observation data. The present ensemble of models incorporates all four combinations tested across 20 individually fitted models, as described in the Table 11. In particular, four fitting procedures were carried using all observation data but varying the vulnerabilities (simulations 1-4 and 11-14). In additional six procedures the fitting was optimised for: i) groups presenting major SS values (simulations 5 and 15), ii) benthic groups (simulations 6 and 16), iii) demersal fish groups (simulations 7 and 17), iv) benthic predators (simulations 8 and 18), v) low trophic level energy grazers (simulations 9 and 19) and vi) top predators (simulations 10 and 20). One last simulation (21) was added to test fitting the model to all observation data estimating all vulnerabilities at once. The fitting procedure was done iteratively, starting the fitting with the predefined initial condition and subsequently adding observation data of similar functional groups or increasing the number of vulnerability parameters until including all the data and all the individual vulnerability parameters. The maximum vulnerability value was limited to 10000 since higher values did not significantly improve the overall fit.

Table 11. Scheme used for fitting the basic Ecosim model for both fitting strategies.

FGs' observation data selected for initial fitting	Predator-prey strategy (P/P)	Predator only strategy (P/O)
All observation data	1. 160 vulnerability values	11. all predators
All observation data	2. 120 vulnerability values	12. 20 predators
All observation data	3. 80 vulnerability values	13. 10 predators
All observation data	4. 40 vulnerability values	14. initially benthos with all predators, later all obs. data with 80 vuln. values
Major SS (Benthiv, Cephal)	5. 80 vulnerability values	15. all predators
Benthos (Bivalv, AnnWorm, DecaSto, Gastrop, Echinod, OthBenth)	6. 80 vulnerability values	16. all predators
Demersal fish (BenthPel, InvFeed, Flatfish, Benthiv)	7. 80 vulnerability values	17. all predators
DecaSto, Cephal	8. 80 vulnerability values	18. all predators
Energy grazers (SmallPel, Herbiv, Bivalv, AnnWorm, OthBenth)	9. 80 vulnerability values	19. all predators
Top predators (MarBird, Elasm, BenthPel, PelPisc, DemPisc)	10. 80 vulnerability values	20. all predators
All observation data	21. all vulnerability values	

6.1. Additional measures of ensemble's goodness of fit

The goodness of fit of all models of the ensemble was estimated using different error measures, allowing analysing different aspects, not all of them present in the standard EwE software. The EwE's built-in sum of squared log deviations (SS) was used as a standard measure of goodness for comparison purposes. Additional measures, the sum of squared relative residuals and the rescaled sum of squared relative residuals were used to estimate the deviation of individual models from the ensemble's mean trend. More detailed information than just SS were obtained using the model's predicted data accuracy, precision and correlation to observed data. The accuracy, indicated as bias, measured the discrepancy of the mean value of predicted data from the observed data, the discrepancy from the desired target, and it might be both positive and negative. The precision, on the other hand, indicated the order of dispersion of predicted values around the mean predicted value, the size of cloud of points around its centre point, measured using the standard deviation. The correlation gave indicated how much the predicted data were in accordance with the observed data.

The sum of squared deviations (SS) using all the observation data was used for general evaluation purposes and for comparison with other EwE models. In each trial, the SS values of each functional groups were grouped into similar macro-functional groups (Tab. 8) to verify the objectives of the fitting strategies and compare to macro group's mean SS.

Each trial of the ensemble was compared with the mean trend of the ensemble ($B_{y,fg,mem}$ vs. $\bar{B}_{y,fg}$) by using the sum of squared percentage residuals (Eq. 32) and the rescaled sum of squared percentage residuals (Eq. 33). The first measures the absolute relative error, penalising trials with large deviations, while the second rescales the trials error based on the largest absolute relative error. The lowest values indicate the model closest to the ensemble's mean trend.

$$SS(\%res)_{cal} = \sum_{fg=1}^{all\,fg}\left[\sum_{y=2005}^{last\,year}\left((B_{y,fg,cal} - \bar{B}_{y,fg})/\bar{B}_{y,fg}\right)^2\right] \tag*{Eq. 32}$$

$$rSS(\%res)_{mem} = \sum_{fg=1}^{all\,fg}\left[\frac{\sum_{y=2005}^{2030}\left((B_{y,fg,mem} - \bar{B}_{y,fg})/\bar{B}_{y,fg}\right)^2}{\max\limits_{mem=1:all}\left(\sum_{y=2005}^{2030}\left((B_{y,fg,mem} - \bar{B}_{y,fg})/\bar{B}_{y,fg}\right)^2\right)}\right] \tag*{Eq. 33}$$

Two different transformations were used to determine the accuracy (the bias) and precision (standard deviation) of the functional groups and make them comparable between different functional groups and between biomass and catch simulations. These transformations were: a) the relative difference of predicted and observed data (Eq. 34) and b) the normalized predicted and observed data values (Eq. 35, 36). Both transformations returned a pure number without units and allow to compare different simulations as they are. The purpose of the first transformation (Eq. 34) is to give an easy-to-understand idea about the ensemble's discrepancies from the observed value, possibly for stakeholders or fisheries management purpose. It measures the relative error of predicted from observed data, i.e. the discrepancy relative to individual observed values (point-to-point). Instead, the normalized transformation (Eq. 35, 36) was used to determine the predicted data bias and precision compared to observed data variation and the correlation between these data. The later compares the predicted data with the mean of observed data and standardizes their variation (point-to-mean value).

$$P'_{y,fg} = \frac{Pred_{y,fg}}{Obs_{y,fg}} - 1 \tag*{Eq. 34}$$

$$P\_norm_{y,fg,type} = \frac{Pred_{y,fg,type} - \overline{Obs_{fg,type}}}{\sigma\left(Obs_{fg,type}\right)} \tag*{Eq. 35}$$

$$O\_norm_{y,fg,type} = \frac{Obs_{y,fg,type} - \overline{Obs_{fg,type}}}{\sigma\left(Obs_{fg,type}\right)} \qquad \text{Eq. 36}$$

Where $Pred_{y,fg}$ and $Obs_{fg,type}$ are the predicted and observed data of selected type (biomass or catch), $P'_{y,fg}$ are the relative biomass or catch data and $P\_norm_{y,fg,type}$ and $O\_norm_{y,fg,type}$ are the normalized predicted and observed data (biomass or catch).

To aid comprehension in terms of management, the bias and mean absolute error (MAE) of each functional group were estimated using the relative difference of predicted to observed data (Eq. 34).

Functional groups' bias, precision and correlation were investigated for all trials of the ensemble and the ensemble's mean trend using a custom-BPC (bias, precision and correlation) and the Taylor diagrams based on normalized predicted and observed data (Eq. 35, 36). The Taylor diagram uses the unbiased relative mean standard deviation (uRMSD, Eq. 37) as a measure of error of trial's predicted data from the observed data.

In an effort to synthesise the functional group's or model's prediction performance, average bias, average precision and average correlation were calculated. Each simulation contributed with only one value therefore had an equal weight in the averaging process. By doing so the influence of the different number of observation points between different simulations was ignored, since the simulation with the higher number of observation points would have a higher weight and a larger contribution to the end result. Averages were calculated for each functional group when both biomass and catch data were present (Eq. 38, 39, 40, 41). The same procedure was used to determine: a) the average fitting of all biomass simulations, b) the average fitting of all catch simulations and c) the ensemble's average fitting of averages of functional group's biomass and catch simulations.

$$uRMSD_i = \sqrt{\frac{1}{N}\sum_{x=1}^{N}\left[\left(Pred_{x,i} - \overline{Pred_i}\right) - \left(Obs_{x,i} - \overline{Obs_i}\right)\right]^2} \qquad \text{Eq. 37}$$

$$\overline{Bias} = \frac{1}{N}\sum_{i=1}^{N} Bias_i \qquad \text{Eq. 38}$$

$$\bar{\sigma} = \sqrt{\frac{1}{N}\sum_{i=1}^{N}\sigma_i^2 + \frac{1}{N}\sum_{i=1}^{N}\left(Bias_i - \overline{Bias}\right)^2} \qquad \text{Eq. 39}$$

$$\overline{uRMSD} = \sqrt{\frac{1}{N}\sum_{i=1}^{N} uRMSD_i^2 + \frac{1}{N}\sum_{i=1}^{N}\left[\left(\overline{Pred_i} - \overline{Pred}\right) - \left(\overline{Obs_i} - \overline{Obs}\right)\right]^2} \qquad \text{Eq. 40}$$

$$\overline{cor} = \frac{\left(\bar{\sigma}_{obs}^{2} + \bar{\sigma}_{pred}^{2} - \overline{uRMSD^{2}}\right)}{2 \cdot \bar{\sigma}_{obs} \cdot \bar{\sigma}_{pred}} \qquad \text{Eq. 41}$$

6.2. Assessing the ensemble’s goodness of fit

The iterative procedure of fitting the models of the ensemble to observation data is illustrated in the Figure 26. Generally, the model’s goodness of fit improved with each iteration. Fitting models to selected observation data would occasionally produce “spikes” that would temporarily worsen the overall goodness of fit when additional observation time series were added. In reality, the fitting procedure would improve the “partial” fitting for selected functional groups, but would worsen the “overall” fit for all functional groups and therefore these “spikes” are a collateral effect. The inclusion of additional observation time series for each functional group would quickly correct this effect.

The models where all observation time series were included in the fitting procedure had both the best goodness of fit (model 21), the lowest sum of squared relative residuals (model 1) and the rescaled sum of squared relative residuals (model 4) (Table 12). Additionally, the predator-prey strategy (PP) achieved better goodness of fit than the predator-only strategy (PR) (Figure 26 and Table 12).

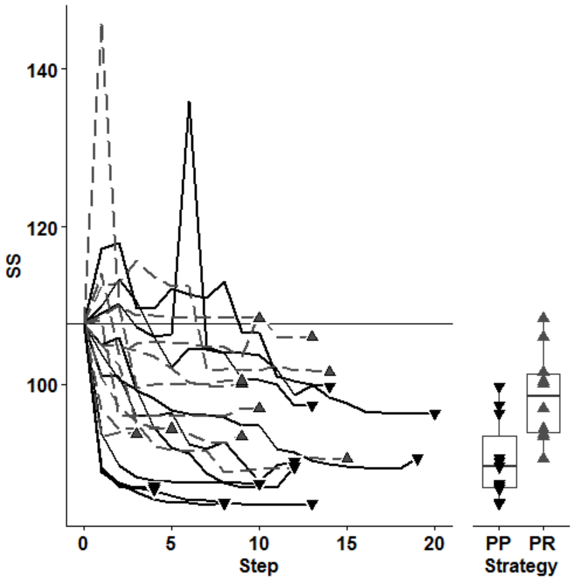

Figure 26. Representation of the iterative procedure of fitting the model to observation data. The goodness of fit is expressed as the sum of squared deviations (SS). The final results of the fittings are represented by triangles, in black for the predator-prey strategy (PP) and grey for the predator-only strategy (PR). The horizontal line is the initial SS.

Reference Ecosim model’s SS: 107.62

Predator-prey strategy (PP)				Predator-only strategy (PR)			
No.	SS	SS%	rSS%	No.	SS	SS%	rSS%
1	86.74	**4.50**	3.40	11	93.67	20.00	8.31
2	87.07	8.45	4.05	12	94.35	23.12	9.05
3	84.96	13.63	5.31	13	93.40	12.23	6.65
4	87.43	4.94	**2.98**	14	90.54	20.41	8.85
5	90.25	7.48	3.79	15	100.16	15.45	9.19
6	96.36	41.63	9.42	16	108.41	17.80	8.38
7	89.55	10.22	6.65	17	100.42	14.92	8.44
8	97.37	12.91	5.56	18	105.96	40.34	13.16
9	99.78	13.03	6.44	19	101.58	17.74	7.98
10	90.61	14.55	10.70	20	96.91	18.00	11.96
21	**84.87**	8.29	4.58				

Table 12. Model sum of squared deviations (SS) after fitting and indicators of sum of squared relative residuals (SS%) and rescaled sum of squared relative residuals (rSS%) against the ensemble’s mean biomass. Lower values are better and the best scores are in bold. The maximum attainable rSS% score in this ensemble was 33.

The effectiveness of the optimised fitting procedure for selected observation data and the fitting behaviour at different number of estimated vulnerabilities are shown in Figure 27. The fitting patterns of macro-functional groups in models 1, 2, 3, 4 and 11, 12, 13 are different between them even though observation data used for fitting are the same. Also, the fitting optimised for selected functional groups managed to meet the objectives in 7 out of 12 models (i.e. models: 8, 9, 10, 16, 17, 18, 19). The opposite, in model 7 the target macro-functional group was penultimate.

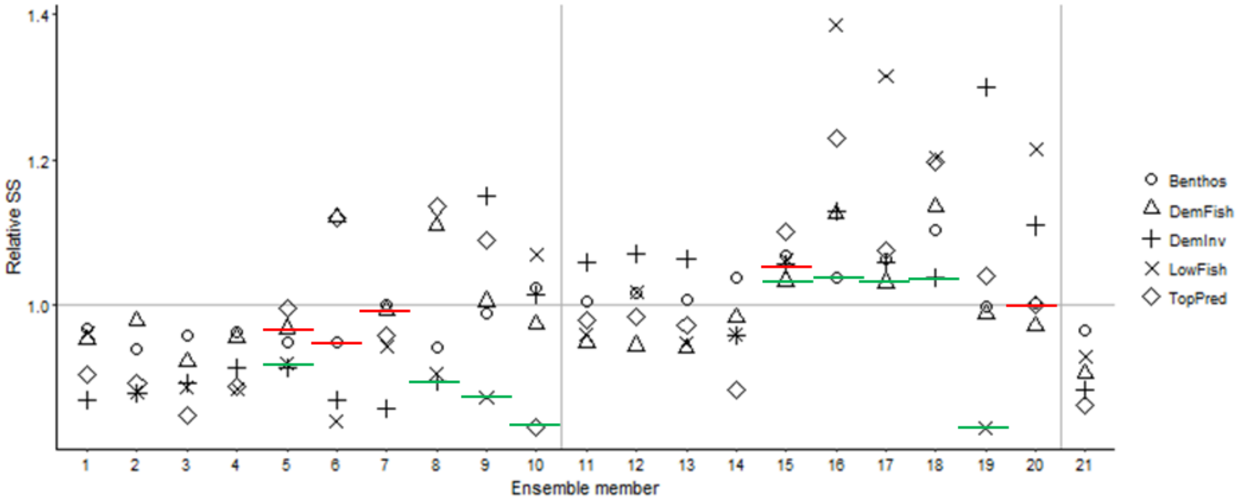

Figure 27. Achieved macro functional groups' fitting for each model in the ensemble. The target macro functional group's relative SS level is indicated by the bars and its fitting performance is green if it is the best performing group, otherwise in red if it is not. Relative SS = $SS_{\text{FGs simulation}} / SS_{\text{FGs ensemble mean}}$, Labels: Benthos: benthic sessile fauna; DemFish: demersal fish; DemInv: demersal invertebrate fauna, LowFish: energy grazing fish (Small pelagic fish, Herbivorous and Detritivorous fish); TopPred: top predators.

The simulations of each model of the ensemble and the influence of different fitting strategies on all functional groups, compared with the observed catch and biomass data when available, are shown in Figure 28 and 29. Models managed to reproduce decently the observed trends of relative biomass data. Catch trends were also reproduced rather well, although for Invertebrate feeding fish, Herbivorous fish, Benthivorous fish, Demersal piscivorous fish and, to a certain degree, for Decapoda & Stomatopoda and Gastropoda the ensemble did not manage to simulate the observed variations. It is also possible to note that simulations based on the predator-prey fitting strategy (PP) are closer to the ensemble's mean trend than are the simulations based on the predator-only strategy (PR).

Finally, Figure 30 shows the influence of the fitting procedure on the variation of the models. During the hindcast the coefficients of variance were lower than in the simulations of the future indicating that the fitting procedure did positively contribute to reproduce the observed trends, except for Suprabenthos, Macro and Micro-zooplankton, Bacterioplankton, Dinoflagellates, POM and Bottom detritus. During the hindcast the largest mean coefficient of variation was observed for Marine birds, Benthopelagic fish, Demersal piscivorous fish and the Cephalopoda, while during the simulated future the largest mean coefficient of variation were held by the Marine birds, Elasmobranchii, Benthopelagic fish, Flatfish and Demersal piscivorous fish.

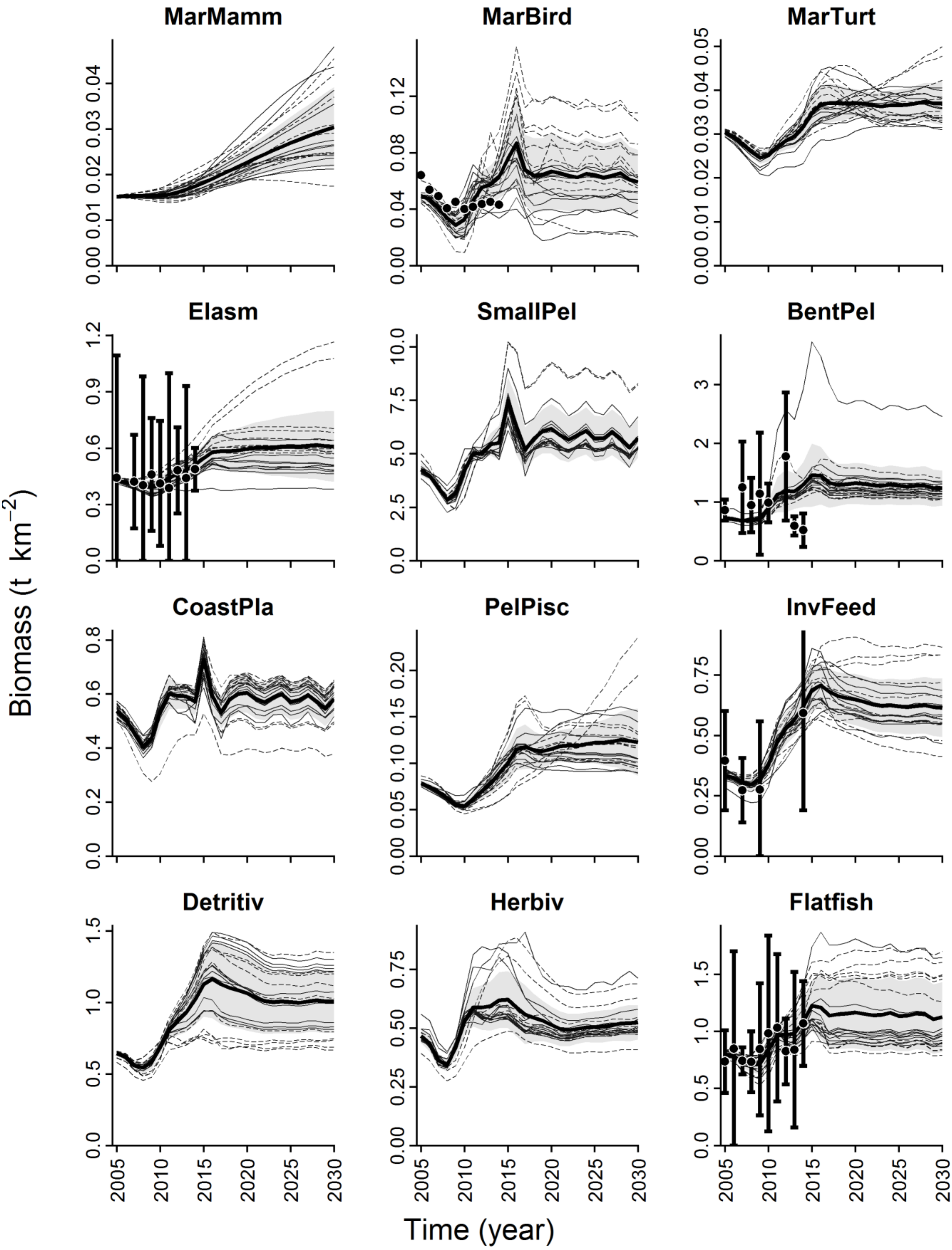

Figure 28. Biomass simulations as a result of different fitting procedures. Model outputs are represented by thin lines; solid line: predator-prey fitting strategy (PP), dashed line: predator-only fitting strategy (PR). Ensemble's mean value and standard deviation are shown as a thick line and the shadowed area. Relative observed biomasses and their standard deviation are shown as dots and error bars. Abbreviations as in Table 6.

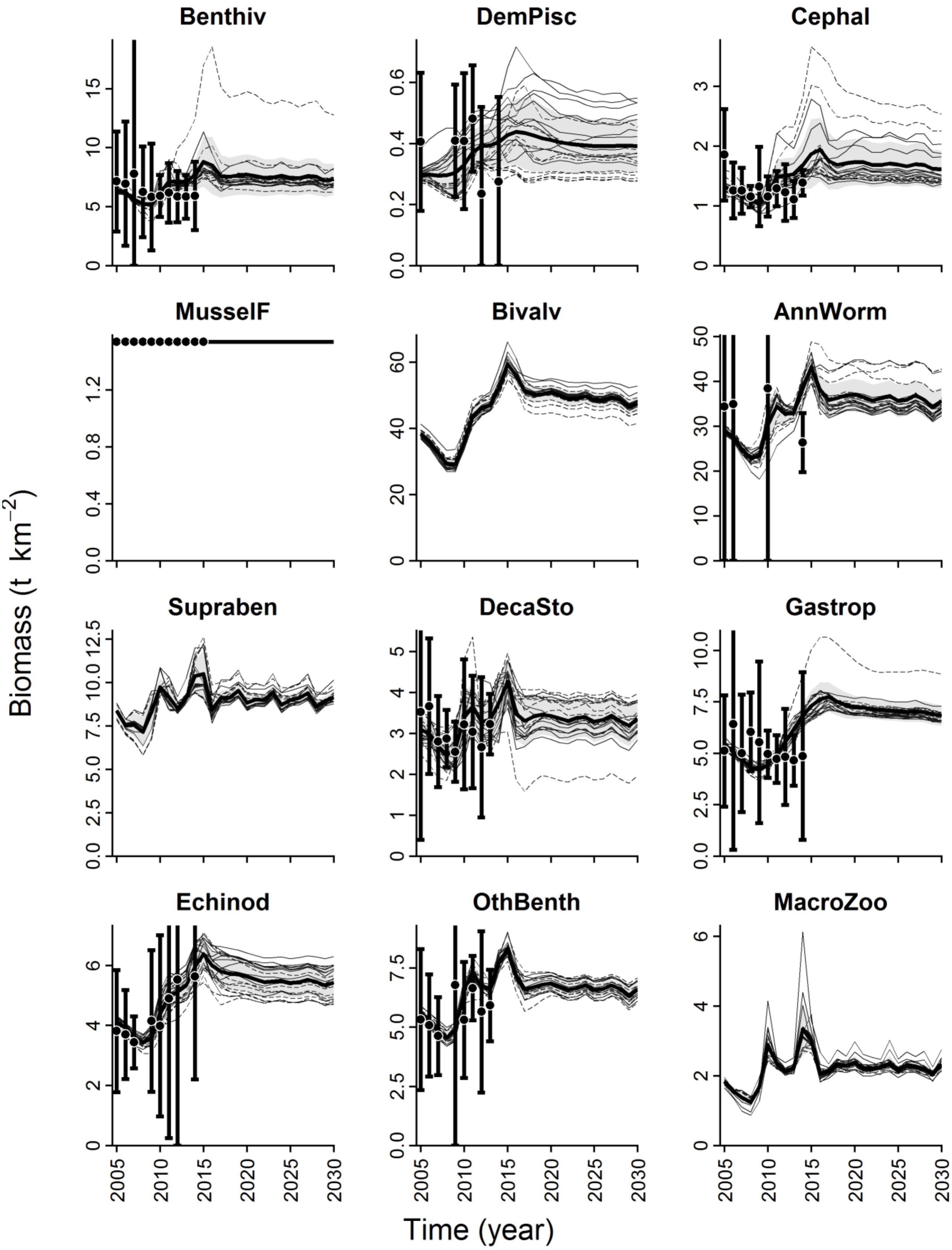

Figure 28 (continue). Biomass simulations as a result of different fitting procedures.

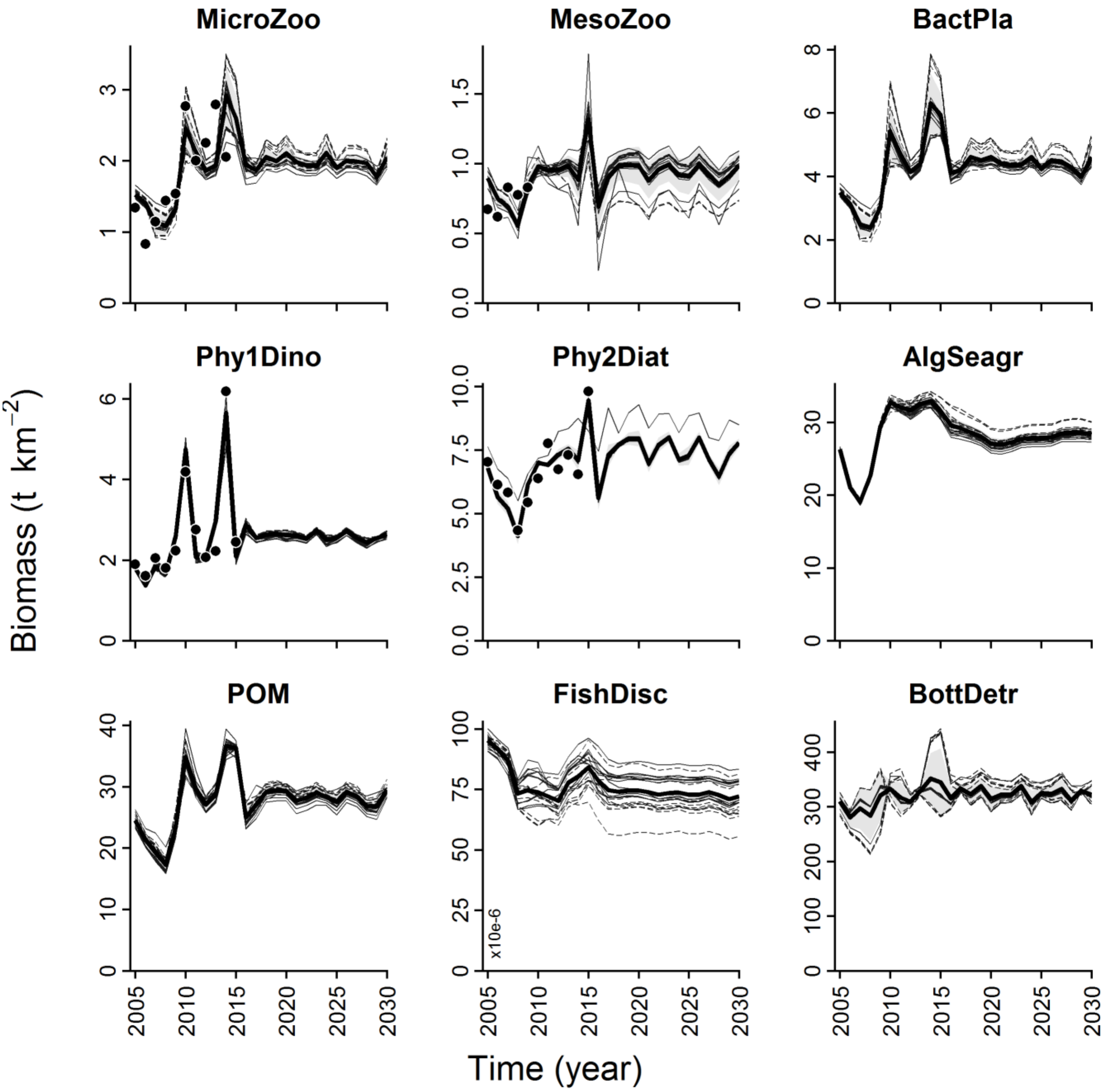

Figure 28 (continue). Biomass simulations as a result of different fitting procedures.

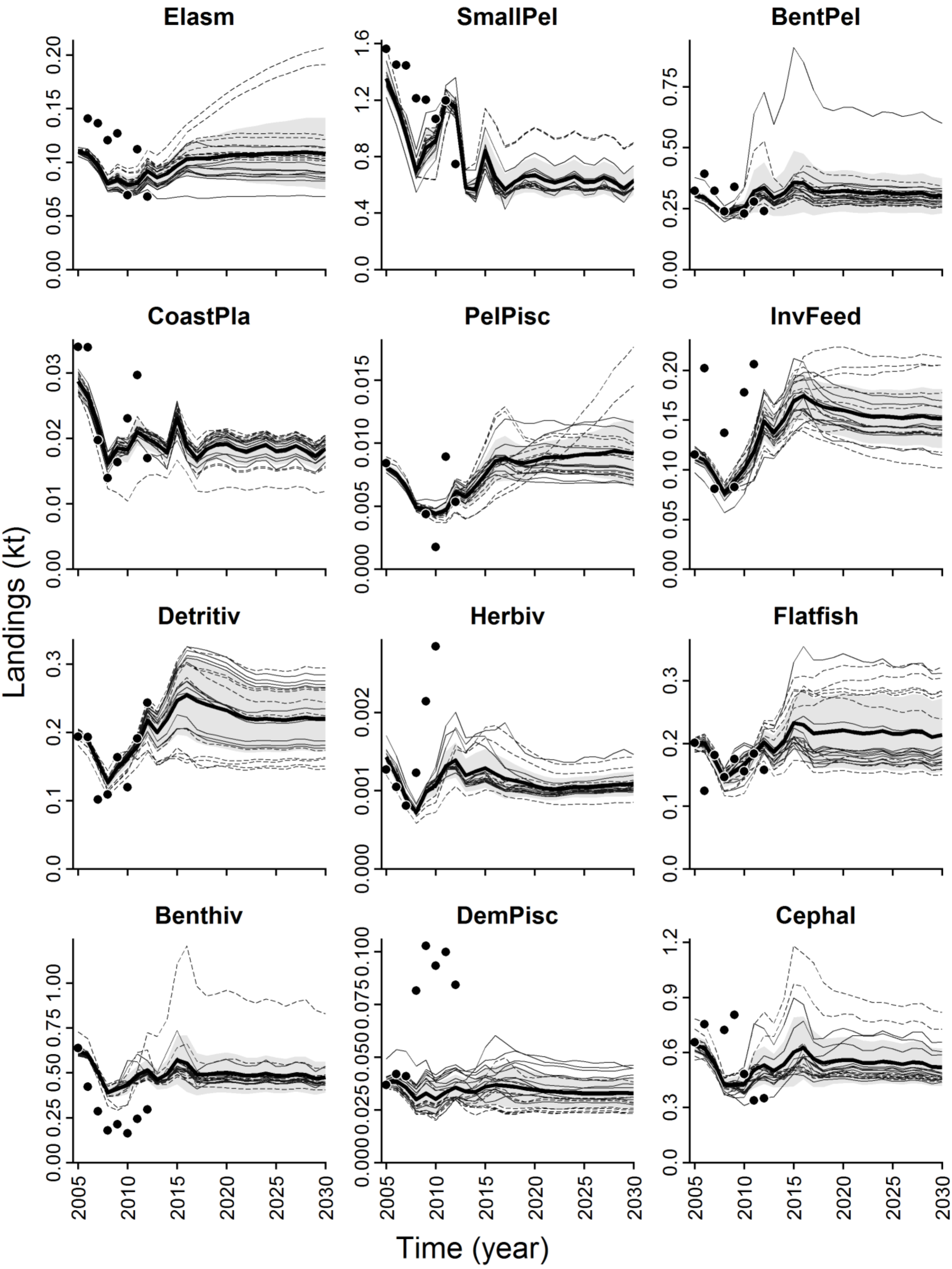

Figure 29. Landings simulations as a result of different fitting procedures. Model outputs are represented by thin lines; solid lines: predator-prey fitting strategy (PP), dashed line: predator-only (PR) fitting strategy. Ensemble's mean value and standard deviation are shown as a thick line and the shadowed area. Absolute observed landing data are shown as dots. Abbreviations as in Table 6.

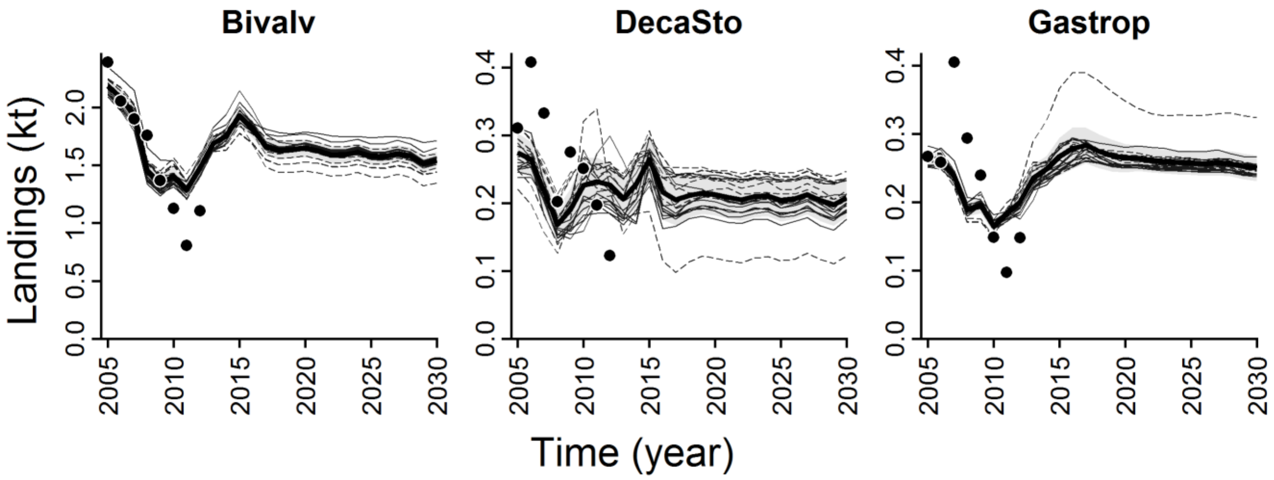

Figure 29 (continue). Landings simulations as a result of different fitting procedures.

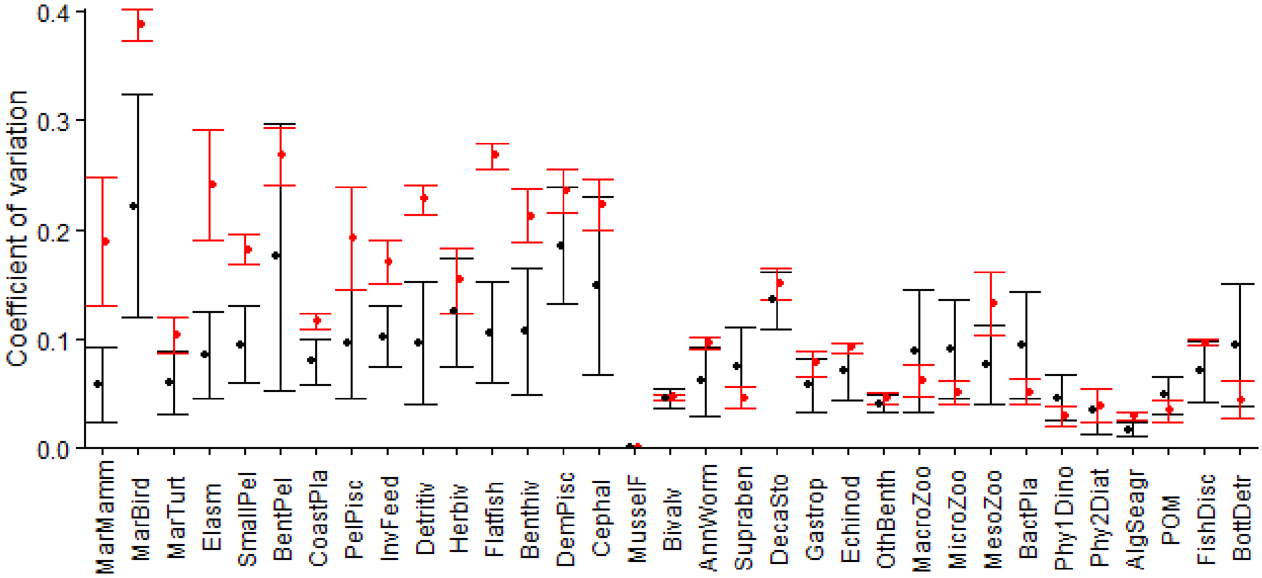

Figure 30. Mean and standard deviation of the coefficient of variation of each functional group. During the hindcast in black and when simulating the future in red. Abbreviations as in Table 6.

6.2.1. Assessing ensemble's bias and precision for management purposes

A representation of relative difference of predicted and observed data bias and mean absolute error (MAE) is illustrated in Figure 31. Biomass simulations (Figure 31.a, Table 13) had both the largest biases and mean absolute errors. Except for Flatfish and Echinodermata, the biomass of all other functional groups was overestimated. In particular, Elasmobranchii, Invertebrate feeding fish, Cephalopoda and Benthivorous fish had both the largest biases and mean absolute errors, respectively: >100%, >150%, >150% and >200%. Echinodermata were the only group centred on the observed biomasses, but with a considerable mean absolute error (93%). Overall, the mean biomass bias resulted in overestimation (73%) and a mean absolute error of 101%.

In contrast, simulated catches (Figure 31.b, Table 13) show both the lowest biases and the mean absolute errors, all lower than 50%, except for the Benthivorous fish. Mean catches are centred on the observed values and have a mean absolute error of 26%.

Overall, mean values for each functional group (Figure 31.c, Table 13) indicate most of the functional groups have biases and error lower that 50%. Again, the Benthivorous fish and Cephalopoda depart largely from the rest of the groups, while Invertebrate feeding fish and Elasmobranchii are much closer to the ensemble's mean bias and absolute error. The Demersal piscivorous fish, Echinodermata and Coastal planktivorous fish have the lowest biases, while the Detritivorous fish, Bivalvia and Coastal planktivorous fish have the lowest mean absolute errors. Finally, the ensemble slightly overestimates the predicted values (bias=32%) and its mean absolute error is 60%.

Table 13. Summary of the bias and mean absolute error (MAE) of each functional group and model's overall. Represented the averages and standard deviations (S.D.) of errors of biomass, catches and average of comparison points of biomass and catch simulations together (Combined). The model's overall (Overall) averages all the comparison between predicted and observed data. Abbreviations of functional groups (FG) as in Table 6.

	Relative absolute error (bias)						Mean absolute error (MAE)					
	Biomass		Catches		Combined		Biomass		Catches		Combined	
FG	Average	S.D.	Average	S.D.	Average	S.D.	Average	S.D.	Average	S.D.	Average	S.D.
MarBird	31%	±48%			31%	±48%	41%	±40%			41%	±40%
Elasm	111%	±248%	-14%	±27%	56%	±196%	133%	±236%	28%	±11%	87%	±185%
SmallPel			-13%	±27%	-13%	±27%			25%	±15%	25%	±15%
BenthPel	64%	±150%	-12%	±17%	26%	±113%	91%	±136%	16%	±13%	53%	±103%
CoastPla			-5%	±21%	-5%	±21%			20%	±9%	20%	±9%
PelPisc			25%	±68%	25%	±68%			46%	±56%	46%	±56%
InvFeed	173%	±186%	-22%	±27%	49%	±147%	184%	±175%	30%	±18%	86%	±130%
Detritiv			10%	±24%	10%	±24%			19%	±18%	19%	±18%
Herbiv			-20%	±34%	-20%	±34%			33%	±21%	33%	±21%
Flatfish	-43%	±26%	9%	±24%	-20%	±36%	44%	±26%	17%	±19%	32%	±27%
Benthiv	203%	±179%	75%	±47%	146%	±151%	211%	±169%	76%	±46%	151%	±146%
DemPisc	70%	±100%	-43%	±30%	6%	±89%	78%	±94%	44%	±27%	59%	±67%
Cephal	176%	±116%	-10%	±42%	93%	±129%	177%	±115%	35%	±24%	114%	±112%
Bivalv			9%	±24%	9%	±24%			16%	±20%	16%	±20%
AnnWorm	58%	±128%	-2%	±26%	18%	±82%	82%	±115%	20%	±17%	41%	±74%
DecaSto	49%	±60%	-2%	±36%	25%	±56%	57%	±53%	27%	±23%	43%	±44%
Gastrop	59%	±138%	4%	±37%	34%	±109%	99%	±113%	27%	±25%	67%	±93%
Echinod	-1%	±93%	8%	±18%	4%	±67%	65%	±67%	15%	±13%	40%	±54%
OthBenth	25%	±120%	4%	±23%	14%	±87%	68%	±102%	17%	±16%	43%	±77%
Overall	**73%**	**±152%**	**0%**	**±40%**	**32%**	**±111%**	**101%**	**±134%**	**28%**	**±28%**	**60%**	**±98%**

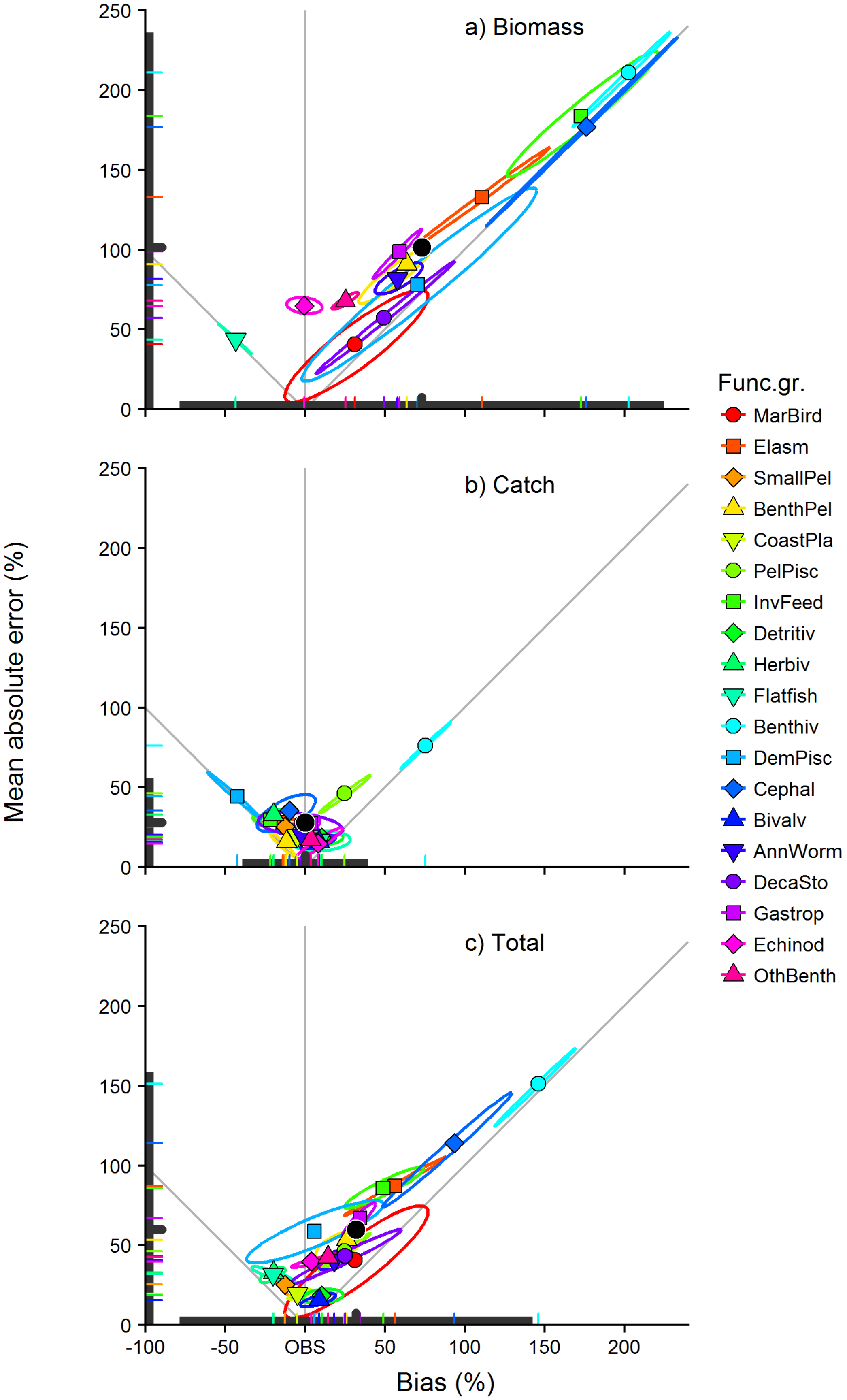

Figure 31. Summary of the bias and mean absolute error (MAE) of each functional group. Functional groups are represented by symbols of different shapes and colours. The ellipses represent the 95% confidence level based on functional groups' individual trials, which for simplicity were omitted. The average of all functional groups is represented by the black point; the standard deviation is represented by the black line along the axis. Abbreviations as in Table 6.

Figure 32 illustrates how different fitting strategies were able to reproduce the observed data. Ensemble's mean values are the same as in Figure 31. In biomass simulations (Figure 32.a) the trials showed a large variability in terms of bias and the absolute error. In particular, the trials 1, 2, 3, 4, 5 and 21 had their simulations the closest to the observed values, while the most distant ones were the trials 6, 9, 16, 18, 19 and 20. On the other hand, in terms of catches (Figure 32.b) the trials had a range of close to 8% for bias and a much closer range (less than 4%) for the absolute error. In this case, the trials 3, 4 and 21 were the closes to the observed values. In terms of overall fitting (Figure 32.c) the pattern is similar as in simulations of biomasses.

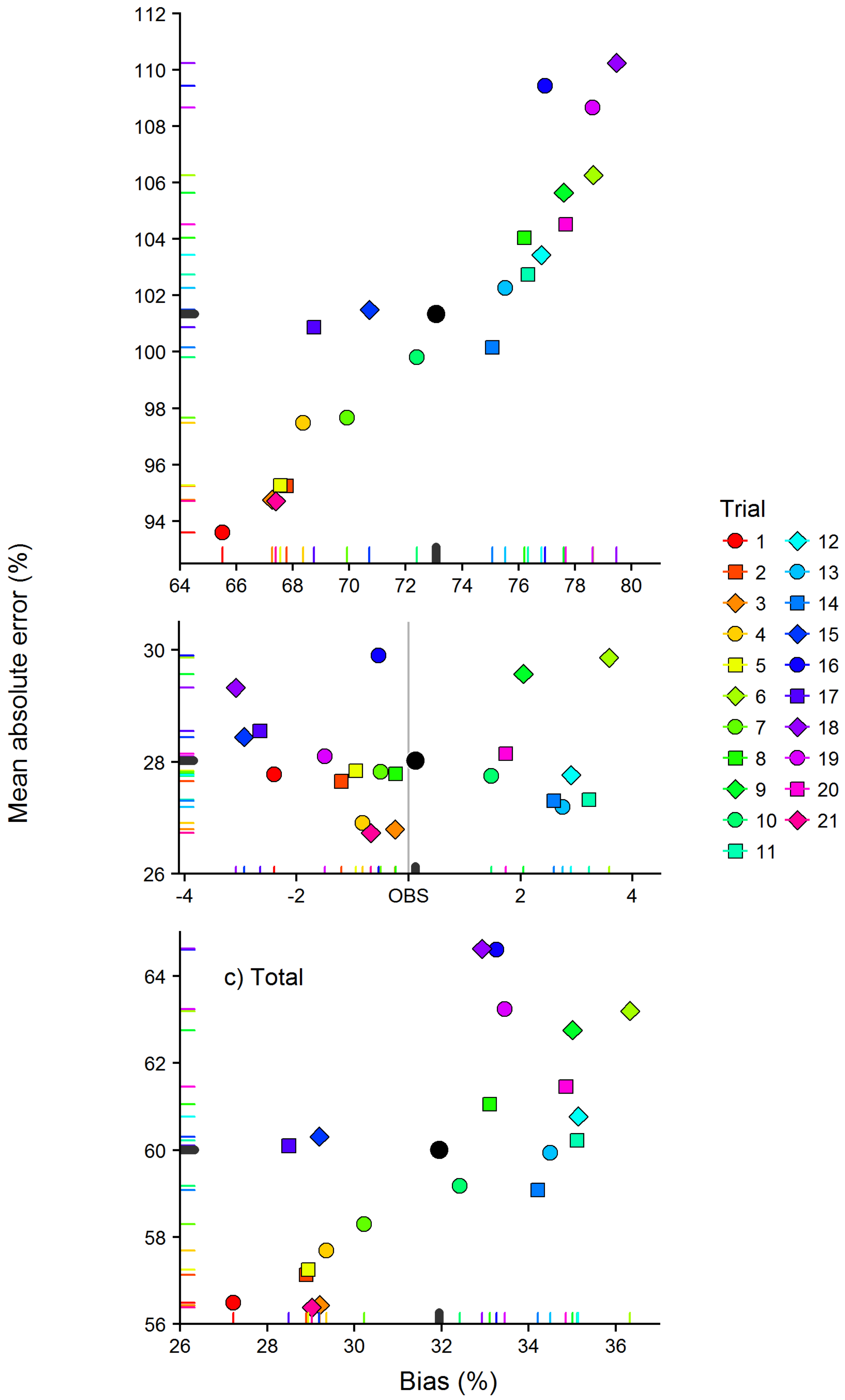

Figure 32. Summary of the mean bias and absolute error of each trial. Each trial is represented by pointers of different shapes and colours. To simplify the figure, the single trials points were omitted. The average of all trials is represented by the black point; standard deviation is omitted because it is out of scale.

6.2.2. Assessing ensemble's bias, precision and correlation

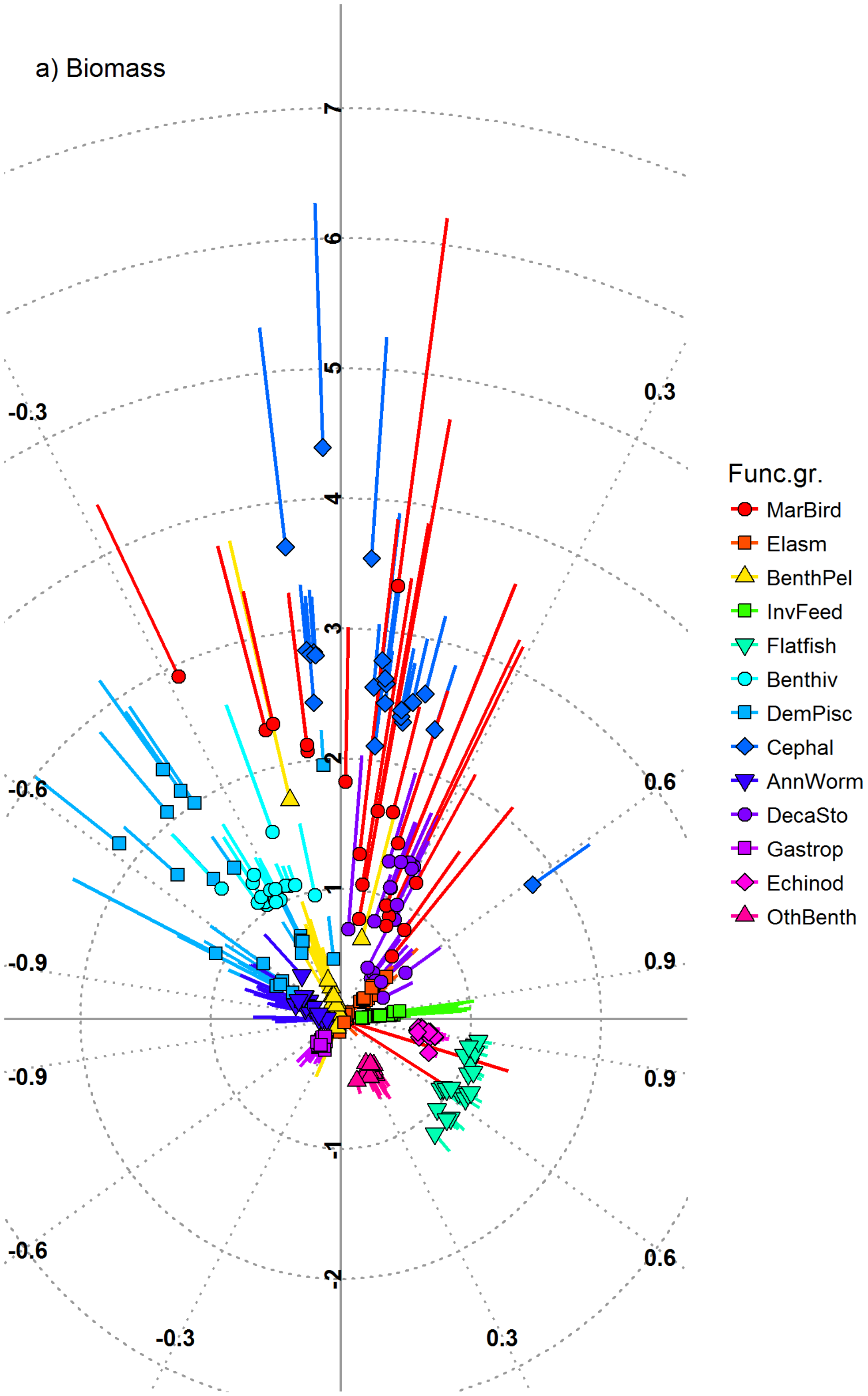

Figure 33. Bias, precision and correlation (BPC) diagram of all trials for the biomass of each functional groups hindcast. Functional groups are represented by symbols of different shapes and colours. The point indicates the bias from the mean observed data and is represented as the distance from the origin, (positive bias above the x-axis, and negative below), the precision (standard deviation) is represented by the length of the lines, and the correlation is indicated by the angle (positive on right, negative on left). The circles represent the scale expressed in observation data standard deviation units. Radial lines are correlation guidelines. Abbreviations as in Table 6.

Individual representation of the bias, precision and correlation with observed data of each functional group for biomass and catches are illustrated respectively in the Figures 33 and 34. The predicted biomasses exhibit a complex situation with functional groups having both low biases, high precision and high correlation (Invertebrate feeding fish, Echinodermata, Flatfish and Elasmobranchii) and the opposite (Marine birds, Cephalopoda, and Demersal piscivorous fish). In particular, it is possible to note how fitting procedures can produce highly variable outputs, ranging from uncorrelated simulations with large biases, to correlated data centred on the observed values (Marine birds). Biases lower than one observed data standard deviation were found in 7 functional groups, and additional 3 were on the borderline (Flatfish, Decapoda & Stomatopoda and Benthivorous fish). Moreover, in catch simulations (Figure 34) most of the biases were lower than on standard deviation of their respective observed data and had mostly positive correlations. Exceptions are the Demersal piscivorous fish (resulting largely negatively correlated), Flatfish (uncorrelated but with low bias), Benthivorous fish (correlated but with larger bias) and the Benthopelagic fish with intermediate positive correlation but large bias.

Different representation of the same normalized data is represented by Taylor diagrams in Figures 35 and 36. A similar situation to Figure 33 is found for the biomasses (Figure 35.a). Most simulations had standard deviation of predicted data smaller than the one, while Marine birds and Cephalopoda had larger discrepancies (Figure 35.a). Catch simulations (Figure 35.b) also had most of predicted data with standard deviation smaller than one, with positive correlations and lower uRMSD values (closer to the observed data on the right).

The Taylor diagram in Figure 36 represents the fittings for individual fitting strategy. The figure reproduces the same patterns observed in Figure 32, but allows seeing the different responses for individual functional groups. The clearest example of different fitting strategies are the symbols equivalent for Marine birds (compare to Figure 35), where trials 3 and 10 had the fitting closest to observed data. To notice that trials with lowest SS or relative errors (1, 4 and 21) are not the closest one to the observed data.

The ensemble's mean trend bias, precision and correlation are expressed in the Figures 38 (biomass), 39 (catch) and 40 (combined biomass and catch) and Table 14. Mean trends of biomass of eight functional groups had the bias and the precision lower than one observation data standard deviation (ODSD), and Flatfish, Benthivorous fish and Demersal piscivorous fish with a bias close to on 1 ODSD (Figure 38, Table 14). The highest correlation (larger than 0.5) were by Invertebrate feeding fish, Echinodermata, Flatfish and Elasmobranchii. However, since the functional groups had mixed responses the mean fitting of biomasses had an uncorrelated trend, with a bias of 0.44 ODSD

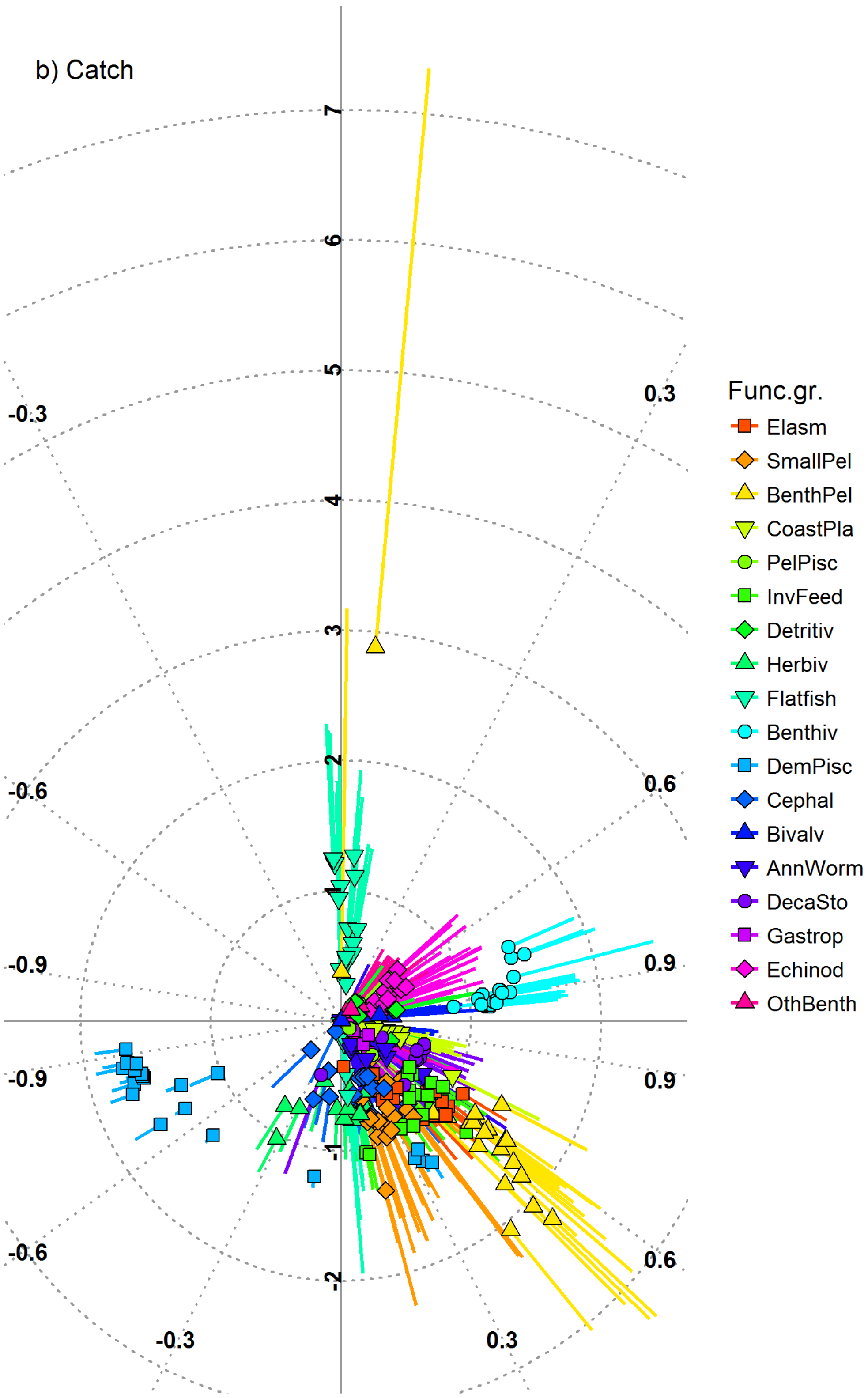

Figure 34. Bias, precision and correlation (BPC) diagram of all trials for the catches of each functional groups hindcast. Functional groups are represented by symbols of different shapes and colours. The point indicates the bias from the mean observed data and is represented as the distance from the origin, (positive bias above the x-axis, and negative below), the precision (standard deviation) is represented by the length of the lines, and the correlation is indicated by the angle (positive on right, negative on left). The circles represent the scale expressed in observation data standard deviation units. Radial lines are correlation guidelines. Abbreviations as in Table 6.

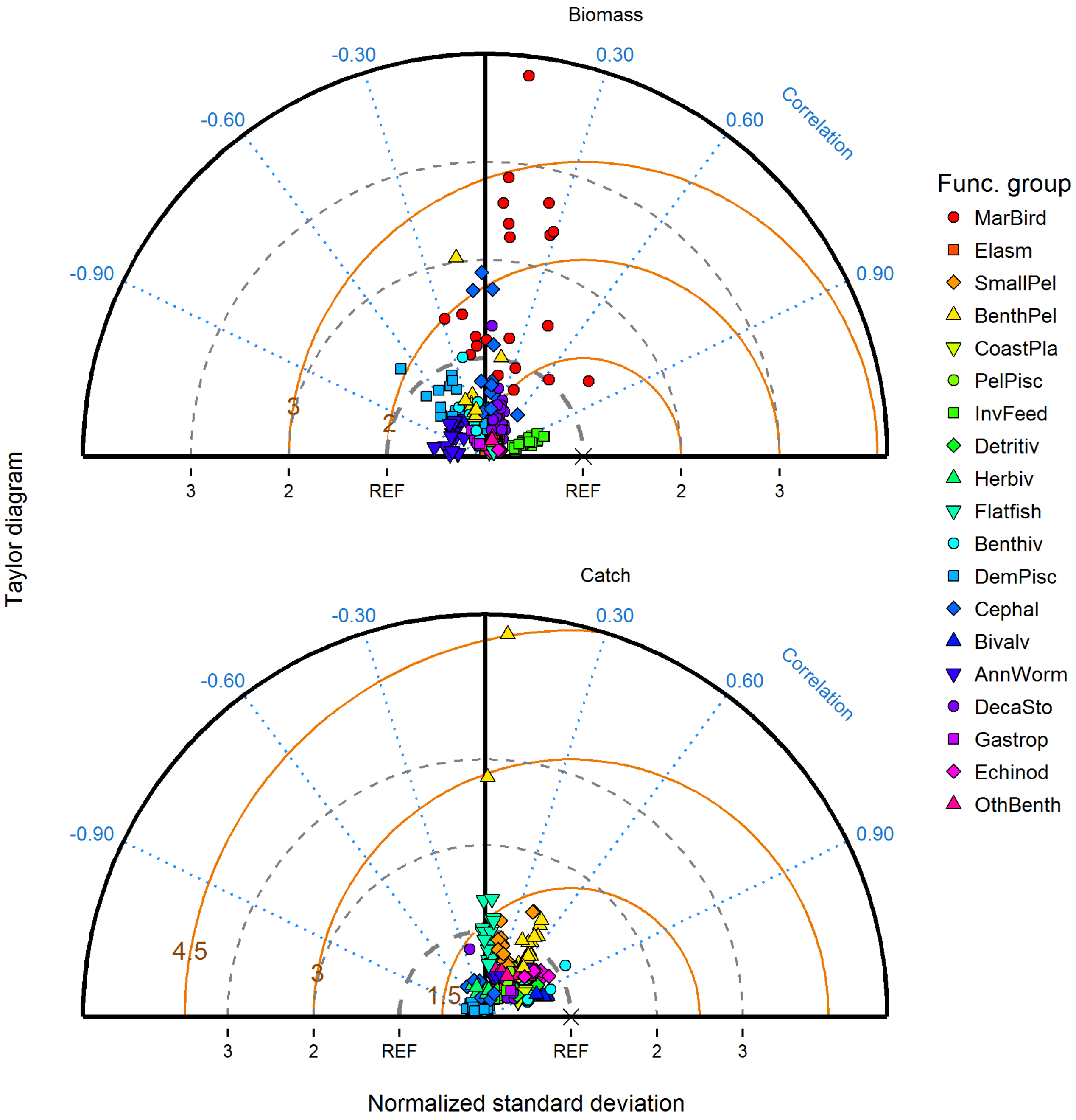

Figure 35. Taylor diagrams of the variation in biomass and catch of each functional group at the end of simulations in the test scenarios. The pointer's distance from the origin indicates the standard deviation of predicted data (grey lines); the observed data standard deviation is represented by the thick grey line as reference (REF); the distance from the observed data (REF with a cross) indicates the unbiased relative mean standard deviation (uRMSD) of predicted data from observed data (orange lines); the angle formed between the pointer and the origin indicates the correlation od predicted data with observed data (blue lines). Abbreviations as in Table 6.

and a precision of 1.12 ODSD. On the other hand, ensemble's mean trend of catches (Figure 39, Table 14) indicated mostly a positive correlation pattern, where only the Demersal piscivorous fish were negatively correlated with the largest bias. Additionally, except for Benthivorous fish and Benthopelagic fish, other functional groups had a bias lower than 1 ODSD, of which nine had a bias

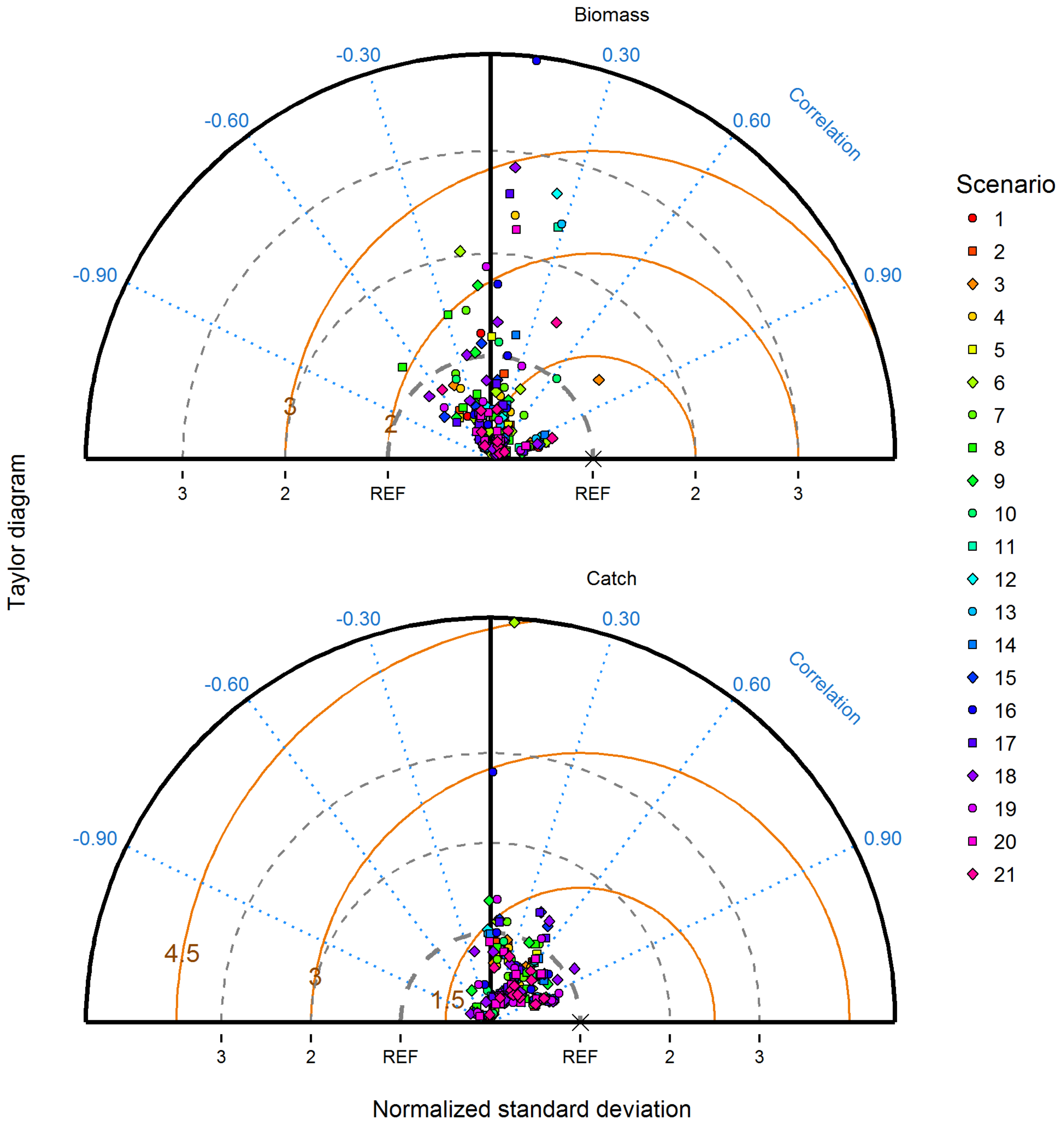

Figure 36. Taylor diagrams of the variation in biomass and catch of each functional group at the end of simulations but grouped for different fitting procedure trials (written Scenario in legend). The pointer's distance from the origin indicates the standard deviation of predicted data (grey lines); the observed data standard deviation is represented by the thick grey line as reference (REF); the distance from the observed data (REF with a cross) indicates the unbiased relative mean standard deviation (uRMSD) of predicted data from observed data (orange lines); the angle formed between the pointer and the origin indicates the correlation od predicted data with observed data (blue lines).

lower than 0.5 ODSD. The overall trend of catches had a low bias (-0.30 ODSD), a precision of (0.85 ODSD) and a positive correlation (0.32). Finally, combining both the mean trends of biomasses and catches (Figure 40, Table 14) allowed to have a complete view of how the individual functional groups performed. Except for Marine birds, Benthivorous fish and Cephalopoda, all had a bias smaller than 1 ODSD, and except for Marine birds, also their precision is smaller than 1 ODSD. The

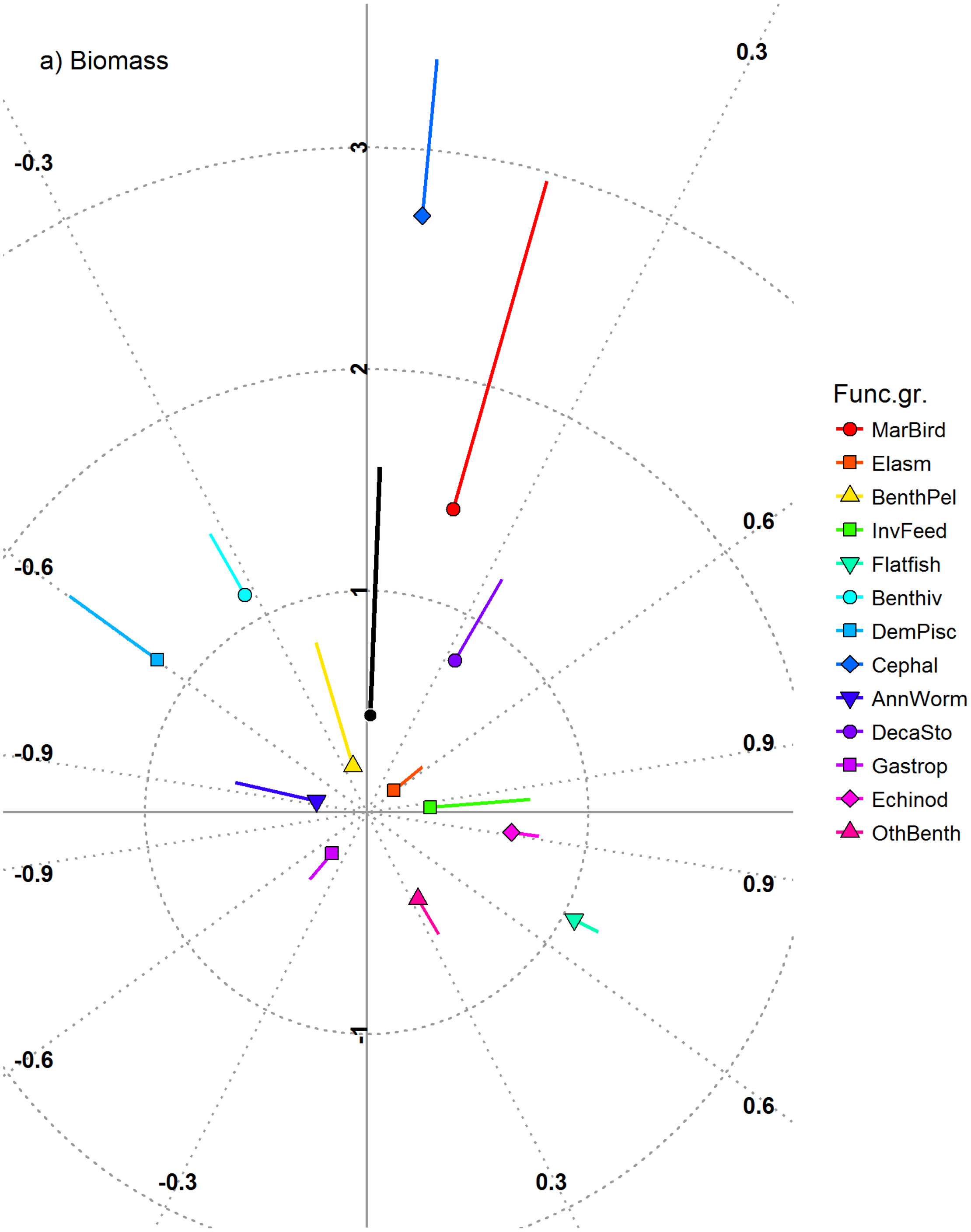

Figure 38. Bias, precision and correlation (BPC) diagram of the biomass of each functional group in all trials in hindcast. Functional groups are represented by symbols of different shapes and colours; the mean is represented in black. The point indicates the bias from the mean observed data and is represented as the distance from the origin, (positive bias above the x-axis, and negative below), the precision (standard deviation) is represented by the length of the lines, and the correlation is indicated by the angle (positive on right, negative on left). The circles represent the scale expressed in observation data standard deviation units. Radial lines are correlation guidelines. Abbreviations as in Table 6.

functional groups' correlations were from uncorrelated to highly positive correlation; only the Demersal piscivorous fish had a negative correlation. In its complexity, the whole ensemble's mean trend was accurate (bias close to zero), had a precision of 1.04 ODSD and low positive correlation (0.19).

Taylor diagrams of biomasses and catches (Figure 41, Table 14) show the same results as described above, but from the perspective of the observed data. For biomass simulations, the Invertebrate feeding fish had the lowest uRMSD, close to 0.5. On the other hand, in catch simulations the lowest uRMSD values were observed for Bivalvia and Benthivorous fish, followed by Detritivorous fish, Coastal planktivorous fish and Echinodermata.

Table 14. Summary of bias, precision and correlation of ensemble's mean trend for each functional group and ensemble's overall averages. The represented values are the number of observations, bias, precision (standard deviation), unbiased relative mean standard deviation (uRMSD) and correlation of predicted data in biomass, catch and the average of combined biomass and catch simulations. Bias, precision and uRMSD are expressed in observation data standard deviation (ODSD) units after normalization of all data (mean of observed data equal to 0 and standard deviation to 1). Functional groups have equal weight in averaging. The abbreviations of functional groups (FG) as in Table 6.

	Biomass					Catches					Averages combined				
Groups	Number of observations	Bias (predicted)	Precision – S.D. (predicted)	uRMSD	Correlation	Number of observations	Bias (predicted)	Precision – S.D. (predicted)	uRMSD	Correlation	Number of observations	Bias (predicted)	Precision – S.D. (predicted)	uRMSD	Correlation
MarBird	10	1.42	1.54	1.68	0.18						10	1.42	1.54	1.68	0.18
Elasm	9	0.16	0.17	0.92	0.57	7	-0.78	0.36	0.89	0.46	16	-0.31	0.55	1.02	0.24
SmallPel						8	-0.82	0.82	1.10	0.29	8	-0.82	0.82	1.10	0.29
BenthPel	8	0.22	0.58	1.25	-0.18	8	-1.24	0.99	1.01	0.48	16	-0.51	1.09	1.35	0.17
PelPisc						8	-0.31	0.54	0.63	0.83	8	-0.31	0.54	0.63	0.83
Coastpla						5	-0.06	0.52	0.86	0.51	5	-0.06	0.52	0.86	0.51
InvFeed	4	0.29	0.45	0.59	0.95	7	-0.87	0.27	0.89	0.52	11	-0.29	0.69	0.95	0.42
Detritiv						8	0.17	0.54	0.60	0.85	8	0.17	0.54	0.60	0.85
Herbiv						6	-0.71	0.31	1.04	0.03	6	-0.71	0.31	1.04	0.03
Flatfish	10	-1.05	0.12	0.92	0.70	8	0.52	0.89	1.31	0.04	18	-0.27	1.01	1.38	0.06
Benthiv	10	1.12	0.32	1.14	-0.33	8	1.21	0.56	0.52	0.93	18	1.17	0.46	0.89	0.46
DemPisc	6	1.17	0.49	1.35	-0.60	8	-1.46	0.12	1.09	-0.76	14	-0.15	1.36	1.80	-0.14
Cephal	10	2.70	0.71	1.19	0.06	8	-0.55	0.27	1.00	0.13	18	1.08	1.71	1.96	0.02
Bivalv						8	0.09	0.69	0.44	0.93	8	0.09	0.69	0.44	0.93
AnnWorm	4	0.23	0.37	1.34	-0.86	8	-0.27	0.53	0.89	0.46	12	-0.02	0.52	1.16	-0.07
DecaSto	9	0.79	0.42	0.95	0.33	8	-0.35	0.40	0.79	0.67	17	0.22	0.70	1.04	0.29
Gastrop	10	-0.24	0.15	1.08	-0.45	8	-0.23	0.42	0.80	0.64	18	-0.23	0.32	0.95	0.32
Echinod	8	-0.66	0.13	0.89	0.91	8	0.24	0.71	0.69	0.73	16	-0.21	0.68	0.91	0.47
OthBenth	8	-0.45	0.19	0.95	0.34	8	-0.03	0.56	0.88	0.48	16	-0.24	0.47	0.94	0.35
Overall	**106**	**0.44**	**1.12**	**1.48**	**0.02**	**137**	**-0.30**	**0.85**	**1.08**	**0.32**	**243**	**0.00**	**1.04**	**1.30**	**0.19**

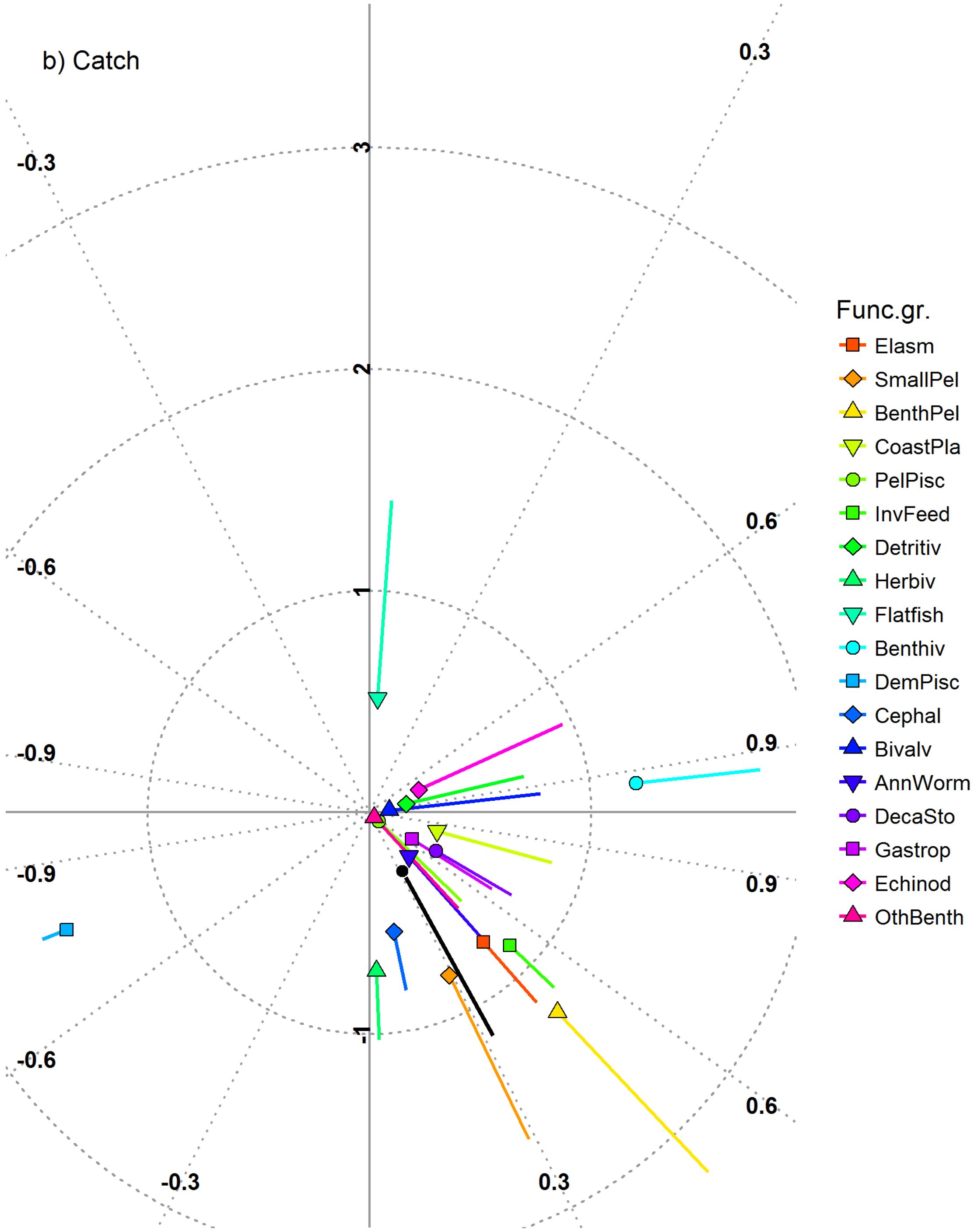

Figure 39. Bias, precision and correlation (BPC) diagram of the catch of each functional group in all trials in hindcast. Functional groups are represented by symbols of different shapes and colours; the mean is represented in black. The point indicates the bias from the mean observed data and is represented as the distance from the origin, (positive bias above the x-axis, and negative below), the precision (standard deviation) is represented by the length of the lines, and the correlation is indicated by the angle (positive on right, negative on left). The circles represent the scale expressed in observation data standard deviation units. Radial lines are correlation guidelines. Abbreviations as in Table 6.

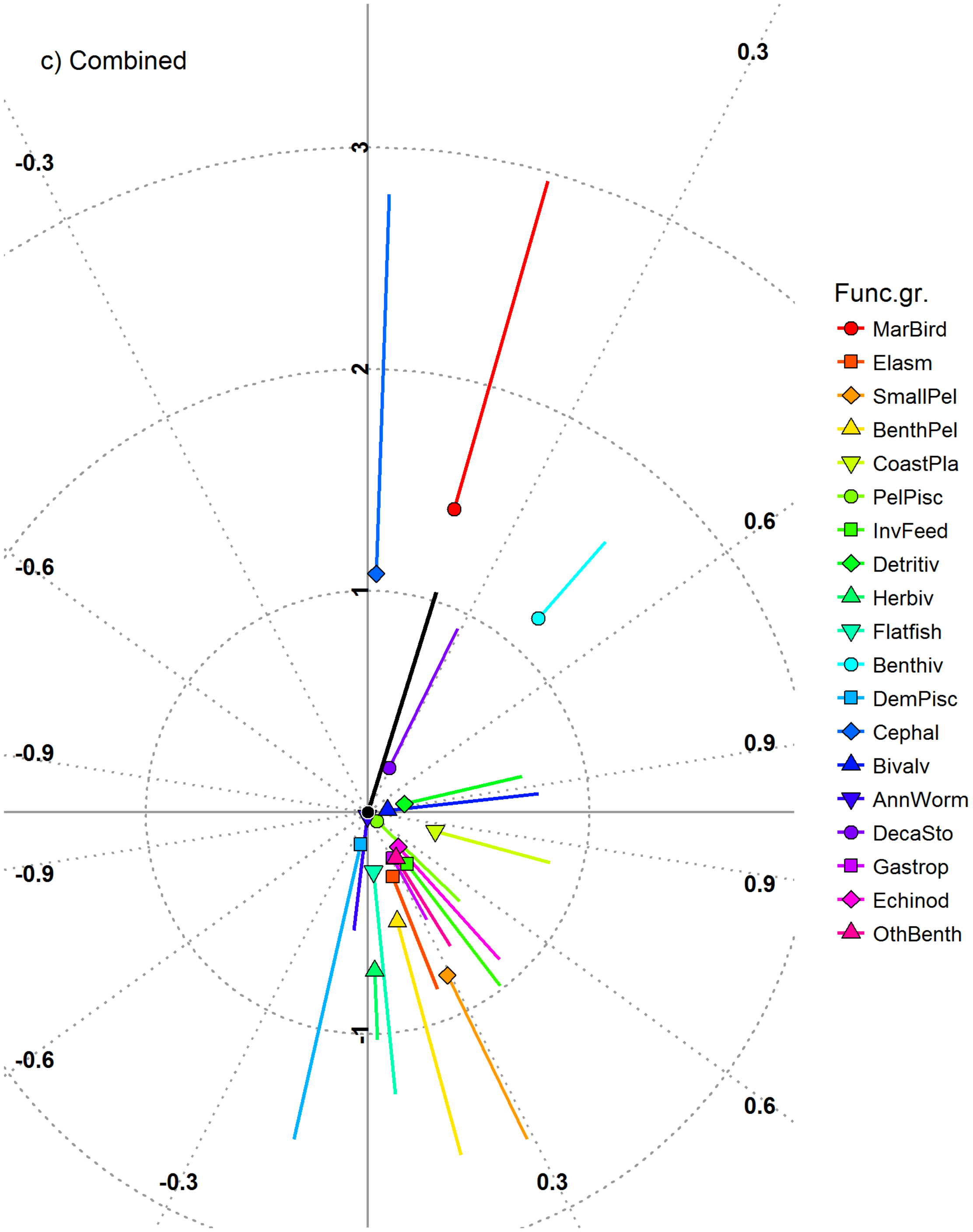

Figure 40. Bias, precision and correlation (BPC) diagram of the mean fitting combining biomass and catch data for each functional group in all trials in hindcast. Functional groups are represented by symbols of different shapes and colours; the mean is represented in black. The point indicates the bias from the mean observed data and is represented as the distance from the origin, (positive bias above the x-axis, and negative below), the precision (standard deviation) is represented by the length of the lines, and the correlation is indicated by the angle (positive on right, negative on left). The circles represent the scale expressed in observation data standard deviation units. Radial lines are correlation guidelines. Abbreviations as in Table 6.

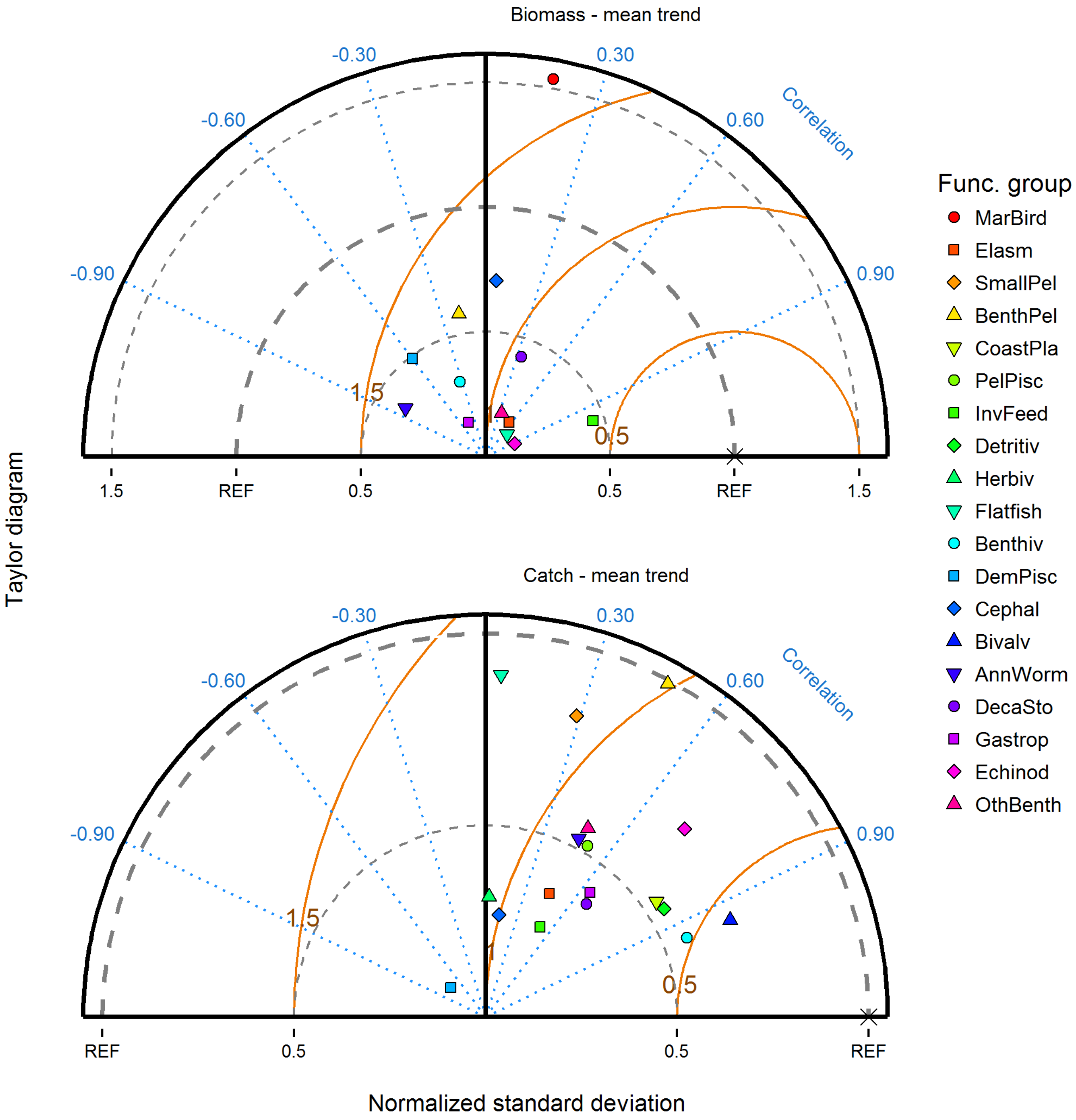

Figure 41. Taylor diagrams of functional groups' biomass variation at the end of simulations in the test scenarios. The pointer's distance from the origin indicates the standard deviation of predicted data (grey lines); the observed data standard deviation is represented by the thick grey line as reference (REF); the distance from the observed data (REF with a cross) indicates the unbiased relative mean standard deviation (uRMSD) of predicted data from observed data (orange lines); the angle formed between the pointer and the origin indicates the correlation od predicted data with observed data (blue lines). Abbreviations as in Table 6.

6.3. Discussion

The development of an ensemble of models based on different fitting procedures and strategies allowed creating multiple simulations that predict the possible trends of different functional groups and assess the uncertainty of different trophic interactions. An in-depth analysis allowed having indications on the ensemble's accuracy, precision, correlation and goodness of fit in reproducing field data.

The results highlight strengths, weaknesses and the difficulties in developing an ensemble of models. The ability to create simulations of trophic responses and to leave the mass-balanced model unchanged is the strengths of using trophic interactions for model fitting. Creating an ensemble of alternative models simulating trophic uncertainty is labour intensive and time consuming since no automated routines are present in the EwE suite, only the stepwise fitting procedure for testing alternative hypotheses and model fitting has been integrated (Scott et al., 2016). Moreover, the use of selected observation data in fitting procedures allows estimating the model's intrinsic trophic uncertainties, even when no other parameter uncertainty data are available. Major differences between fitting strategies were also highlighted (Figure 26, Table 12): better smoothness in sequential fitting when all observations are used and higher fitting power occur when more vulnerability values are used. In terms of vulnerability, selecting individual predator-prey interactions, as in the PP strategy, allowed to estimate the vulnerability values that will better fit the predicted simulations to observed data. In terms of observation data, the best results were achieved when all the observation data were used in the fitting process. Although it is not advised to use a single nonlinear optimisation routine to estimate all available vulnerabilities at once (Christensen et al., 2008), in the present ensemble this fitting strategy, model 21, produced the lowest SS values (Table 12). This result might indicate that the non-linear optimization routine can still handle parameters in medium complexity models, like the present one with 33 functional groups. On the other hand, the selection of observation data for specific functional groups does not guarantee the predicted simulations for those groups will fit better than other functional groups (Figure 27). Even if the observation data are the same, the fitting pattern changes when different setup of vulnerabilities is used for model fitting (Figure 27). Models using the predator-prey strategy (PP) were closer to mean ensemble's trend and exhibited smaller variation between simulations than predator-only simulations (PR) because the use of individual vulnerability values allowed to better balance the fitting of all functional groups simultaneously (Figure 28, Table 12). Also, the lowest bias from observed data was in models fitted using the combination of estimating individual vulnerabilities with all observed data (Figure 32). This result suggests the trophic interactions have a strong impact on the variability of simulations and should also be accounted for when assessing model's simulation uncertainty.

Overall, the ensemble of models was able to reproduce the trends of observed relative biomasses and absolute catch data. The variability during simulations of trophic interactions uncertainty (different fitting strategies, Figure 30, up to 40%) was always larger than the variability during simulations with changes of initial parameters (Monte Carlo, Figure 22.a, up to 5%). Even so, the simulated trophic uncertainty was still smaller than the uncertainty of observed biomass data, and the simulations were contained inside that range (Figure 28).

The effect of different fitting strategies was highlighted in the difference between predicted simulations and observed data for each functional group (Figure 31, 33, 34 and 35). Either based on relative or normalised predicted data indicators, the bias, the variance of the error and the correlation with observed data varied considerably, both for biomass and catch simulations. For certain functional groups that variation would range from accurately and correlated predictions to largely biased and uncorrelated predictions (Figure 33, 34 and 35).

In the NEAS model, based both on relative or normalised predicted data indicators, the simulated catch predictions were more accurate, more correlated and more precise than biomass predictions (Figure 31, 33, 34 and 35). Biomass simulations had a tendency to be overestimated and uncorrelated (Figure 31.a, 33) while catches had positively correlated trend with most biases smaller than 50% or within one standard deviation of the observed data (Figure 31.b, 33). However, macro-benthic fauna was predicted more accurately and with smaller errors than the highly mobile species like the Cephalopoda, fish and Marine birds. The present situation where catch predictions are more accurate than biomass prediction was observed also in the Atlantis model of southern Benguela (Ortega-Cisneros et al., 2017), however, the NEAS model's predictions had a lower correlation. Such discrepancies may be due to different sampling effort of the observed data, the suitability and catchability of the sampling gear for the sampled species or the effect of the model's open boundary and migration of the species. Although the MEDITS survey uses the otter trawl, which is designed for sampling demersal fauna, the sampling effort should be intensified to better reproduce the interannual variations of highly mobile species and the timing and the position of the survey should be more constant to improve the attainability of the sampled data. On the other hand, the SOLEMON survey is carried in the same period, the sampling stations are constant and it uses the rapido trawl to target mainly benthic species, and since many are sessile that sampling scheme and effort may be enough for reliable biomass estimations. The opposite, catch data had the highest sampling rate since they are based on official commercial landings statistics (monthly), therefore were able to integrate better the internannual variation.

The mean trend of the ensemble displayed the same general pattern as functional groups in single simulations, but in many cases reducing the error (Figure 38, 39 and 41). The variability of bias, precision and correlation of each functional group notwithstanding, the overall mean simulations of biomasses and catches were close to the observed data, but catch predictions had higher positive correlations. Finally, the combined predictions of functional groups indicated the ensemble of models was actually accurate (Figure 40) or slightly overestimating the predictions (Figure 31.c), with a good precision and low positive correlation (Figure 40).

The proposed NEAS ensemble of models was able to give insights into possible uncertainties related to trophic interactions and their scale. Because of different trophic interactions, functional groups' predictions had a variable accuracy within the ensemble. However, accuracy and precision are not the only criteria for evaluating models (Starfield, 1997) and the analysis of the model uncertainty can still prove useful in generating policy options that are robust to that uncertainty (Essington, 2004). The present NEAS ensemble managed to predict the major observed biomass and catch trends with different degrees if accuracy and contributed to estimate part of the model related uncertainty, both useful to obtain more reliable management evaluations.

7. *Simulation of effects of the Landing obligation in the NEAS*

In this chapter Landing Obligation issues will be explored with a quantitative approach by using the ensemble of models, obtained from fitting procedures in Chapter 6, including also fisheries dynamics and revenues. Ecosystem models allow to analyse direct and indirect effects of fisheries policies on complex dynamics and to quantify their ecological and economic implications on medium and long terms (see for example Walters and Martell, 2004; Moutopoulos et al., 2013). The ensemble of calibrated models will be used to simulate the gradual application of landing obligation from 2015 to the full application in 2019, as required by the regulation (EU, 2013). Results are reported in terms of differences in biomass at sea, total catches and discards, marketable and non-marketable landings as well as their revenues with and without the LO. The aim of this chapter is to quantify the role of the discards in the ecosystem and provide a comprehensive socio-ecological quantitative framework of possible effects of the LO in the North-eastern Adriatic Sea as exemplificative for Mediterranean mixed fisheries.

7.1. The role of discards in the ecosystem

To evaluate the role and the effects of discards removal form the ecosystem due to the LO, the discards have been divided in two categories: the discards for whom the LO regulation does not apply (C_{od}) and those subject to the LO (C_{LO}). The later were estimated as the discarded proportion of catches for species regulated by the Annex III of the EC 1967/2006 regulation (Table 2 and 15).

Therefore, each fishing gear's total catches (C_{tot}) were divided into three parts: marketable catches (C_m), LO subject catches (C_{LO}) and other discarded catches (C_{od}) ($C_{tot} = C_m + C_{LO} + C_{od}$, follow the Figure 42). Same division was adopted for the total landings (L_{tot}) and the total revenues (R_{tot}). The marketable catches are composed of commercial catches later sold at the fish market for direct human consumption (i.e. $C_m = L_m$, marketable landings), deriving the marketable revenues (R_m). In reference scenario (REF) all dead discards, including those of species with MCRS defined ($C_{od}+C_{LO}$), are discarded at sea, i.e. they flow back in the food web through the functional group Fishery discard. In the implementation of LO (scenario LO), instead, the quantities of discards regulated through MCRS (C_{LO}) are landed for purposes other than direct human consumption ($L_{LO}=C_{LO}$), generating revenues from LO (R_{LO}), while other discarded catches (C_{od}) are returned to the sea to the Fishery discard group. Traditionally the total discard (D_{tot}) consist both of LO subject catches ($D_{LO}=C_{LO}$) and other discarded catches ($D_{od}=C_{od}$) and this classification was maintained even after the introduction of the LO.

Table 15. Fishing catches and aquaculture production (landings and dead discards; t y-1) by fishing gear for the reference year 2005. Only dead discards are reported (calculated as discards multiplied by mortality rates); between parenthesis the proportion of discards subjected to landing obligation. Functional group codes as reported in Table 6. Fishing gears: otter trawlers (OTB), rapido trawl (TBB), mid-water pair pelagic trawl (PTM), purse seines (PS), small-scale fisheries (SSF), dredges (DRB) and mussel farms shell-free wet weight (AQU).

	Landings (C_m) [t y^{-1}]							Dead discards (C_{LO}+C_{od}) [t y^{-1}]; Discards subjected to landing obligation (C_{LO}) [%]						
	OTB	TBB	PTM	PS	SSF	DRB	AQU	OTB	TBB	PTM	PS	SSF	DRB	AQU
Elasm	27.9	0.2	6.3		78.5			0.2						
SmallPel	5.2		715.5	835.3	6.7			172.7 (88.2%)		73.4 (81.8%)	71.0 (100.0%)			
BentPel	126.1		32.1	69.9	94.7			617.4 (45.2%)		0.7 (40.0%)	1.8 (23.2%)	24.7 (66.6%)		
CoastPla	19.3			5.1	9.4			41.3	0.1					
PelPisc			0.5	0.1	7.6									
InvFeed	13.3		0.1	1.1	100.6			7.0 (100.0%)				28.9 (100.0%)		
Detritiv	13.8		10.7	17.9	150.7			0.1		20.9				
Herbiv				0.8	0.4									
Flatfish	1.2	48.2			151.3			2.1	1.7 (15.2%)				4.7	
Benthiv	192.1		14.7	30.7	400.5			475.3 (74.3%)		1.6	0.9 (47.1%)	40.4		
DemPis	14.2	4.0		2.0	16.4			0.7						
Cephal	249.5	51.2		5.5	348.7			24.5	10.8					
MusselF							1586.8							158.6
Bivalv	0.01	252.4			537.7	1599.8		0.01	1154.2				3014.2	
AnnWorm								7.3	7.4				0.3	
DecaSto	63.9	14.5			232.2			28.2	84.1			167.3	110.7	
Gastrop	21.5	172.4			73.3			39.7	31.9			3.5	28.5	
Echinod								473.2	62.9	0.5			1.8	
OthBenth								335.5	129.8					
MacroZoo								0.01						
Total	**748.0**	**542.9**	**779.9**	**968.4**	**2208.7**	**1599.8**	**1586.8**	**2225.2**	**1482.9**	**97.1**	**73.7**	**264.8**	**3160.2**	**158.6**
C_{LO}								**792.4**	**0.2**	**60.4**	**71.8**	**45.5**	**-**	**-**

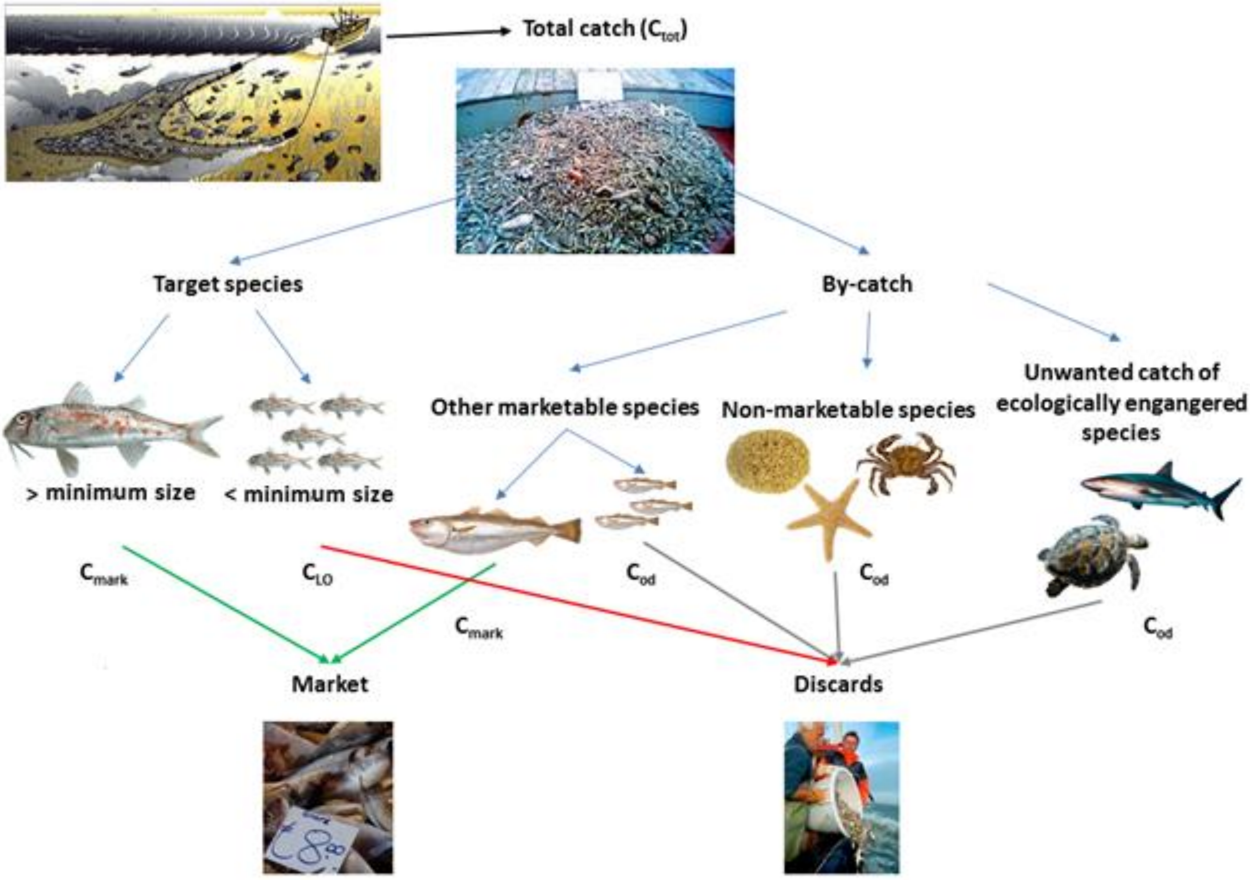

Figure 42. Schematic representation of the division of the catches. In the present scheme the catches are divided according to the Landing Obligation, but the catch of undersized regulated species (C_{LO}) are discarded to the sea, as in the basic scenario.

The setting of additional dummy fleets catching only C_{LO} quantities allowed for a full control of the fate of discards in Ecosim simulations on the basis of the same initial conditions (Ecopath) and the implementation of the scenarios. The management of the fate of LO subject catches (C_{LO}) requires the creation of two dummy fleets (LO_DISCARDING, LO_LANDING) consisting each one in nine (only those subject to LO) of the NEAS simulated fleets (SLO-OTB, SLO-PTM, SLO-PS, SLO-SSF, FVG-OTB, FVG-TBB, FVG-PTM, FVG-LPS, FVG-SSF). In the LO_DISCARDING fleets the discards subjected to landing obligation are thus represented in Ecopath as discards and this fleet is used in the REF scenario. The LO_LANDING fleets simulates the landing of LO subject catches (C_{LO}) by setting the landings equal to 1/1000 of discards subjected to LO, and in the REF scenario its fishing effort is set to zero. The simulation of the LO scenario is done by reducing the LO_DISCARDING fleets relative fishing effort to zero in Ecosim (to simulate no discard at sea) and by increased the LO_LANDING fleets effort from 0 to 1000 multiplied by each fleet's relative fishing effort (to simulate the landing of such discard). A preliminary test of this scheme for representing LO highlighted the perfect correspondence of discarded/landed flows (observed relative differences always $< 10^{-6}$). Landed discards by LO were also assumed to be potentially transformed in fishmeal and thus a price of 0.20 EUR/kg was assigned, roughly 10% of the fishmeal stock price or fishmeal pellets for aquaculture.

The simulation of the landing obligation was performed by decreasing the fishing effort of the LO_DISCARDING fleets while simultaneously increasing the fishing effort of the LO_LANDING fleets by 20% each year from 2015 to 2019. Results of the LO scenario were compared with the REF scenario without the LO.

Although the complete future simulation period was used for comparison among scenarios, synthetic results were obtained by comparing averages of biomasses, landings, discards and revenues of the last 5 years of the simulations (years 2026-2030). According to the Shapiro-Wilk normality test not all data were normality distributed. Significant changes in mean values between scenarios were identified with non-parametric Mann-Whitney U test (*wilcox.test*, R Core Team, 2016), while significant differences between scenarios and reference simulation with the non-parametric ANOVA (*kruskal.test*, R Core Team, 2016): only the significant cases are reported in the following and, if not otherwise indicated, they refer to Mann-Whitney U test.

7.2. Ecologic consequences

In the hindcast period (common to all scenarios and not subjected to any LO), the most important fishing activities in terms of landings are the small-scale fishery (SSF) and the clam dredges

(DRB) (Figure 43). Marketable landings in 2014 accounted for 54.8% (5852 tons) of the total wild catches (excluding discarded live species), while dead discards reached approximately 4662 tons. The mussel farms added approximately another 158 tons (shell-free wet weight) of discards, for a total of 4820 tons. Approximately 583 tons (12%) of discards would be subjected to LO regulation. Notably, although OTB and TBB together provide the main contribution to NEAS discards, only a small portion of their catches is subjected to LO (Figure 43). On the opposite discards from PTM and PS are almost entirely composed of species subjected to LO (i.e. small pelagics; Figure 43). Although data highlighted a decline in catches from 2005 and 2014, their relative proportion among fishing gears remained similar over the analysed period (Figure 43).

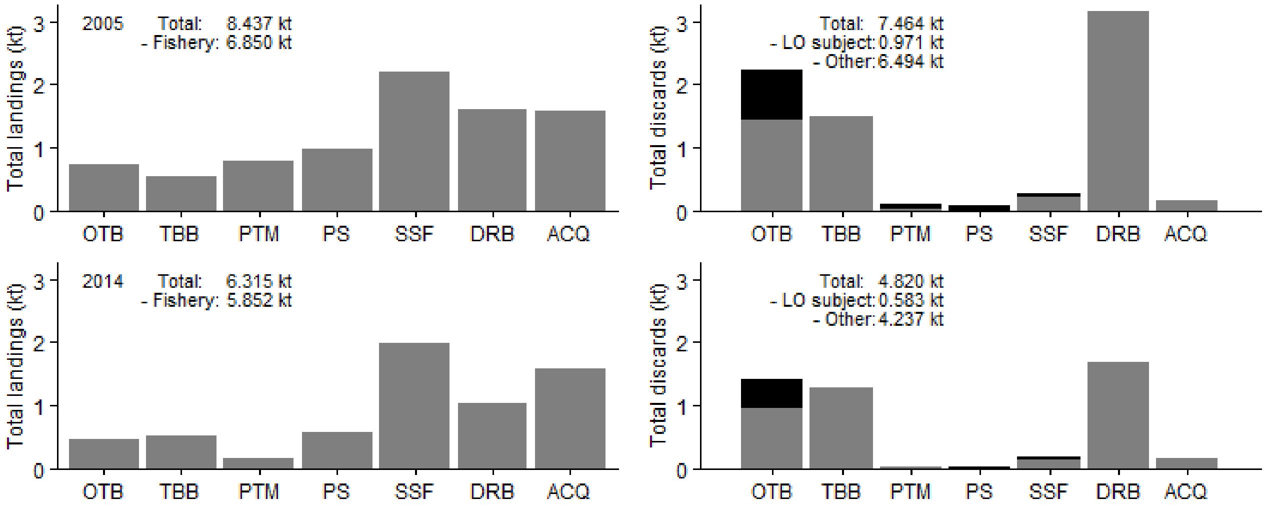

Figure 43. Marketable landings (L_m) and discards (D_{tot}) in the NEAS area at the beginning of the simulations (2005) and prior the application of the LO regulation (2014). Fishing gears: otter trawlers (OTB), rapido trawl (TBB), mid-water pair pelagic trawl (PTM), purse seines (PS), small-scale fisheries (SSF), dredges (DRB) and mussel farms shell-free wet weight (ACQ). The discards are divided in those subject to LO (D_{LO}, black) and others (D_{od}, grey).

In the NEAS model the LO does not impact all functional groups subject to LO the same way. All the discards from Invertebrate feeding fish had to be landed, and high proportions discards from Small pelagic fish and the Benthivorous fish should be landed, while on the other hand, only a small part of Flatfish discards was subject to this regulation (Table 15, 16).

The application of the landing obligation significantly reduced the fishery discards input into the food web with respect to the REF scenario (median of -10.49%, Kruskall-Wallis test p<0.01, Mann-Whitney U test, p<0.01). Moreover, it caused smaller, but significant, differences in mean biomass and catches in several functional groups (Table 16, Figure 44), such as Marine birds (median of -1.51%, p<0.01) Cephalopoda, Demersal piscivorous fish, Benthivorous fish, Decapoda & Stomatopoda, Echinodermata, Flatfish, Benthopelagic fish, Bivalvia, Gastropoda and Elasmobranchii

($p < 0.01$ in all comparisons; Table 1). Conversely, the median biomass of Invertebrate feeding fish and the Small pelagic fish increased significantly (0.56% and 0.13%, respectively, $p < 0.01$ in all comparisons; Table 16).

Table 16. Changes in biomasses, landings and revenues due to the introduction of the landing obligation. Changes are reported as median value (Med%), inter-quartile range (IQR%) of differences between business as usual (REF) and Landing Obligation (LO) scenarios simulated with the ensemble. # Functional groups are reported with their abbreviated code (see Table 6 for full names), Significant changes (bold) emerges from non-parametric Mann-Whitney U test (MW, p<0.05) and Kruskall-Wallis (significant cases at p<0.05 and p<0.01 are highlighted with * and **, respectively).

	Biomass			Total landings (L_{tot})			LO subject landings (L_{LO}/L_{tot_REF})			Total revenues (R_{tot})		
Code$^{\#}$	Med%	IQR%	MW	Med%	IQR%	MW	Med%	IQR%	MW	Med%	IQR%	MW
MarMamm	-0.11	0.31	0.07									
PelPisc	0.03	0.34	0.91	0.03	0.34	0.91				0.03	0.34	0.91
MarBird	**-1.51**	**1.54**	**<0.01**									
DemPisc	**-0.74**	**0.95**	**<0.01**	**-0.74**	**0.95**	**<0.01**				**-0.74**	**0.95**	**<0.01**
MarTurt	0.09	0.32	0.08									
BentPel	**-0.35**	**0.31**	**<0.01**	****84.79**	**0.55**	**<0.01**	****85.14**	**0.28**	**<0.01**	**2.92**	**0.32**	**<0.01**
Elasm	**-0.19**	**0.26**	**<0.01**	**-0.19**	**0.26**	**<0.01**				**-0.19**	**0.26**	**<0.01**
Cephal	**-0.85**	**0.41**	**<0.01**	**-0.86**	**0.41**	**<0.01**				**-0.86**	**0.41**	**<0.01**
Flatfish	**-0.37**	**0.24**	**<0.01**	**-0.23**	**0.24**	**<0.01**	****0.13**	**0.01**	**<0.01**	**-0.36**	**0.24**	**<0.01**
Benthiv	**-0.68**	**0.31**	**<0.01**	****42.97**	**0.46**	**<0.01**	****43.66**	**0.15**	**<0.01**	**0.44**	**0.32**	**<0.01**
SmallPel	**0.13**	**0.15**	**<0.01**	****23.22**	**0.27**	**<0.01**	****23.10**	**0.12**	**<0.01**	***2.18**	**0.17**	**<0.01**
CoastPla	0.04	0.17	0.13	0.04	0.17	0.13				0.04	0.17	0.13
InvFeed	**0.56**	**1.23**	**<0.01**	****31.30**	**1.6**	**<0.01**	****30.74**	**0.37**	**<0.01**	**3.29**	**1.26**	**<0.01**
MacroZoo	0.01	0.19	0.07									
MicroZoo	0.03	0.11	0.16									
Gastrop	**-0.25**	**0.41**	**<0.01**	**-0.25**	**0.41**	**<0.01**				**-0.25**	**0.41**	**<0.01**
DecaSto	**-0.66**	**0.31**	**<0.01**	**-0.66**	**0.31**	**<0.01**				**-0.66**	**0.31**	**<0.01**
Detritiv	-0.02	0.08	0.39	-0.02	0.08	0.39				-0.02	0.08	0.39
OthBenth	-0.04	0.16	0.73									
MesoZoo	<0.01	0.16	0.07									
Herbiv	<0.01	0.03	0.39	<0.01	0.03	0.35				<0.01	0.03	0.35
Echinod	**-0.49**	**0.37**	**<0.01**									
AnnWorm	-0.02	0.23	0.51									
Bivalv	**-0.28**	**1.41**	**<0.01**	**-0.28**	**1.41**	**<0.01**				**-0.28**	**1.41**	**<0.01**
Supraben	0.01	0.09	0.11									
BactPla	0.03	0.11	0.25									
Phy1Dino	**0.07**	**0.09**	**<0.01**									
Phy2Diat	**0.03**	**0.16**	**<0.01**									
AlgSeagr	<0.01	0.04	0.41									
POM	**0.12**	**0.32**	**<0.01**									
FishDisc	****-10.49**	**3.17**	**<0.01**									
BottDetr	**-0.02**	**0.06**	**<0.01**									

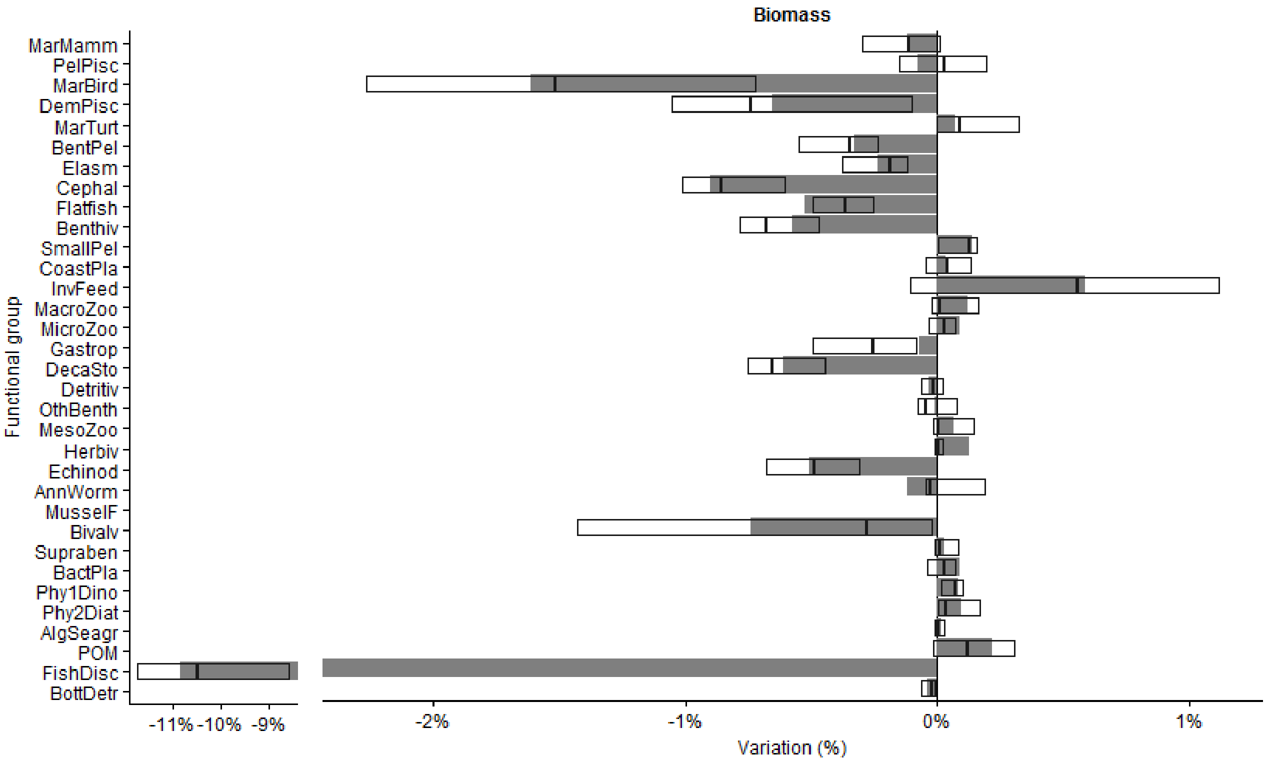

Figure 44. Landing obligation effects on functional groups biomasses. Bars indicate the ensemble's mean value, while box plots the median and the 1st and 3rd quartile. Abbreviations as in Table 6.

Changes in marketable landings and revenues had the same size and statistics as the variation in biomass (omitted from Table 16 because redundant). The largest increase in total landings (that includes the quantities subjected to LO that cannot be sold for human consumption), was observed for the Benthopelagic fish (median 84.8%), followed by the Benthivorous fish, Invertebrate feeding fish and Small pelagic fish (medians 43.0%, 31.3%, 23.2%, respectively; $p<0.01$ for all of them; Kruskall-Wallis test, $p<0.01$; Table 16). For these groups, however, the increase in revenue was much smaller, i.e. from 0.44% for Benthivorous fish to 2.92% for Benthopelagic fish and none for the Flatfish ($p<0.01$ in all comparisons; Table 16).

7.3. Socio-economic consequences on the fishery

Results by gear (Figure 45), highlighted that the LO application would induce significant increase in *total* landings for OTB (median +107.48%, $p<0.01$, Kruskall-Wallis test, $p<0.01$) and, to a lesser extent, for PS, PTM and SSF (median increases by 7.19%, 5.77% and 2.40%, respectively; $p<0.01$ and Kruskall-Wallis test, $p<0.05$ in all comparisons). By contrast, *total* landings for TBB and DRB total landing would decrease (median -0.40%, -0.28%, respectively; all $p<0.01$). However, *marketable* landings for human consumption decreased significantly for DRB, OTB, TBB and SSF (Figure 45; all $p<0.01$). These changes are reflected in the revenues, however, their relative variations are smaller (Figure 45).

When only *marketable* revenues (R_m) are concerned (i.e. in the hypothesis of no revenues for C_{LO}), all fleets have a reduction in revenues with DRB, OTB, SSF and TBB having the worst performances (roughly mean revenues decrease by –0.5%; PS: p=0.53, others p<0.01). Conversely, considering also the hypothetical revenues from C_{LO} (R_m+R_{LO}), the highest increase was attained for OTB (median +2.99%), followed by PS and PTM (median 0.50% and 0.23%, respectively; all p<0.01), while *total* revenues were negative for SSF, TBB and DRB (median -0.43%, -0.44% and -0.28% respectively; all p<0.01) (Figure 45).

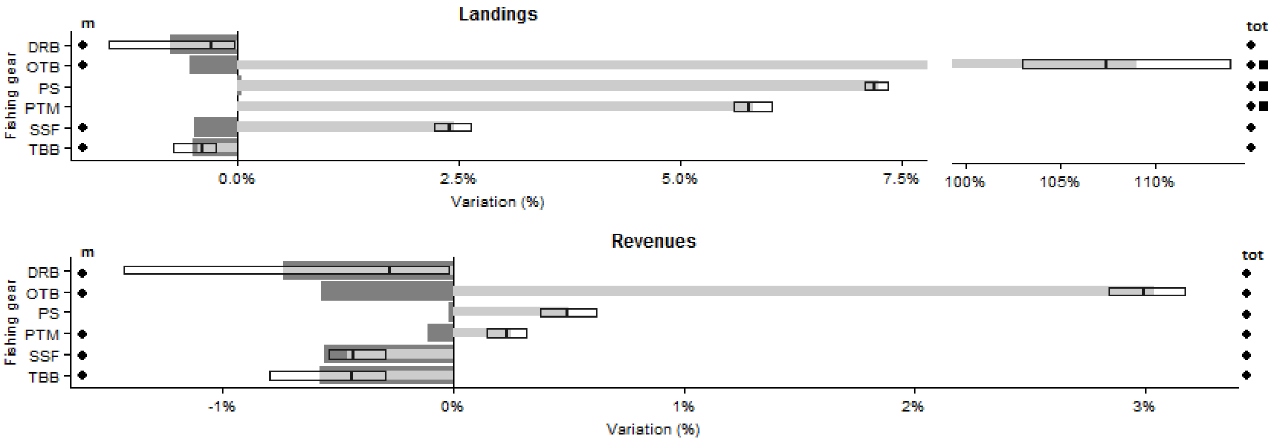

Figure 45. Changes in landings and revenues by fleet as consequences of the introduction of the Landing Obligation. Total landings and revenues (narrow light grey) and marketable contribution (wide dark grey) are disaggregated to highlight the different effects. Box-plots represent median, 1st and the 3rd quartile. Significant differences ($p<0.05$) are highlighted by black dots (Mann-Whitney U test) and black squares (Kruskall-Wallis test) for total (tot) and for marketable landings/revenues (m). Fishing gears: otter trawlers (OTB), rapido trawl (TBB), mid-water pair pelagic trawl (PTM), purse seines (PS), small-scale fisheries (SSF) and dredges (DRB).

Table 17 synthesizes the results of the LO implementation. The LO had an overall significant negative effect on the ecosystem reducing the biomass by a median -0.20% (detritus groups, phytoplankton, bacterioplankton, micro-, meso- and macro-zooplankton were excluded, Table 17). Additionally, although significant negative effects were observed on the fleet's *marketable* landings, the direct effects of the LO regulation would finally lead to a significant increase of *total* landings by 13.30%. Similar significant negative effects are also indicated for the revenues from *marketable* landings (median -0.48%), potentially compensated only by incomes derived from landing the LO regulated discards, assuming they can be sold for other industrial processing.

Table 17. Summarized effects of the landing obligation regulation. Median value (Med%), inter-quartile range (IQR%), non-parametric Mann-Whitney U test p-value (MW). Significant changes ($p<0.05$) are in bold. Biomass effects were calculated by excluding detritus groups, phytoplankton, bacterioplankton, microzooplankton, mesozooplankton and macrozooplankton.

	Total (X_{tot})			Marketable (X_m)			Subject to LO (X_{LO})		
	Med%	IQR%	MW	Med%	IQR%	MW	Med%	IQR%	MW
Biomass	**-0.20**	**0.39**	**<0.01**						
Landings	**13.30**	**1.29**	**<0.01**	**-0.37**	**0.37**	**<0.01**	**13.58**	**1.50**	**<0.01**
Revenues	0.09	0.42	0.33	**-0.48**	**0.37**	**<0.01**	**0.57**	**0.08**	**<0.01**

7.4. Discussion

The actual legal framework defined by the CFP for the implementation of the LO consists in the formulation of discard plans. The plans already issued for the Mediterranean Sea have defined some exemption rules (in particular the “de minimis”) which allows for a phasing in provisions as well as a number of temporal exemptions to the enforcement of the LO for some demersal (Commission Delegated Regulation (EU) 2017/86) and small pelagic (Commission Delegated Regulation (EU) No 1392/2014) fisheries. These exemption rules are granted according to specific thresholds (e.g., the fraction of discards per stock and fishing gear being lower than a threshold, typically 5%-7%) and more importantly, are based on the sector declarations. Therefore, although in the Mediterranean Sea the full application of LO measures will be avoided in the short term, its partial application cannot be dismissed. In particular, for the Adriatic Sea (GSA17 and 18) they limit the application of the LO by considering gear specific *de minimis* exemptions for the anchovy, sardine, mackerel, horse mackerel, hake, red mullet, common sole (EC, 2014; EC, 2017). These exemption rules are granted according to specific thresholds (e.g., the fraction of discards per stock and fishing gear is lower than a threshold, typically 5%-7%) and are based on the sector declarations. Additionally, survivability exemption was also accounted for the common sole (EC, 2017), while for the *Venus spp.* by way of derogation, the MCRS in Italian territorial waters has been reduced to a total length of 22 mm. (EC, 2016).

In this analysis, it is shown that the application of LO would cause a number of changes that, although small, unequivocally indicate that this management measure would have negative impacts on several aspects. In case of a partial application of LO the changes would be smaller, but in the same direction. Assessments based on socio-economic models have already shown that LO will have negative effects on the fisheries sector if not applied in systems regulated by quotas and subjected to strict control (Condie et al., 2014; Batsleer et al., 2016; García et al., 2017). Other analyses based on fisheries-dependent data (Bellido et al., 2017) showed the critical effects of LO for the fisheries sector. However, the use of the quantitative ecosystem approach allows the estimation of long-term impacts of fishery management policies on the marine food web, including trophic cascades, on ecosystem components and fishing activities. Considering the ecological and energetic role of discards, the current work enables to highlight that cascading effects are going to further increase the negative impacts of the LO. Although in this work no test was carried to assess the effects of the application of exemptions rules to LO, that would reduce its absolute effects; the direction of change presented here still hold for a partial LO application.

Incentives to take up selective gears and spatio-temporal effort reallocation are not enough without a rigorous control and enforcement of the regulation (Ulrich 2016). To produce positive effects, the discard ban should be embedded in a wider management system (Condie et al. 2014). However, selective fishing, i.e. avoiding areas or time periods where discards could be caught did not prove to be a sufficient measure for the currents state of European fisheries (Condie et al. 2014) and additional measures, like reducing the by-catch by improving the gear selectivity should be pursued (Campos et al. 2013; Colloca et al. 2013; Batsleer et al. 2016). For a management plan that deals with the discard issue to be effective and have beneficial effects, it should consider adaptive plans that account for fishermen behaviour and the current state of resources (Simons et al. 2015; Hoff and Frost 2016).

The present analysis highlighted that the LO implementation had several significant negative effects: i) on the ecosystem, by reducing the biomass by a median -0.20% (excluding lower TL groups); ii) on the fleet's marketable landings (C_m) that decreased by a median -0.37% and iii) on revenues from marketable landings (R_m) reduced by a median -0.48%. Notably, the LO direct effects include the significant increase of total landings (C_m+C_{LO}) by 13.30%. The potential revenues generated by selling LO quantities (R_{LO}) would only compensate negative changes in revenues from marketable products, resulting in no significant changes in total revenues (Table 17). Notably these results do not consider the additional costs that might emerge from handling (on board and afterwards) the discards landed. Therefore, in reality the economic assessment might be even more negative than in the present conservative estimates.

Most of the direct negative effects of LO involved scavenging species. However, the trophic cascade throughout the food web induced by landing the discards lead to species-specific reactions, even for groups whose diet does not comprise the discards. Indeed, the major variations in the biomass of NEAS functional groups were not observed because of direct effects of LO on scavengers, but because of indirect effects through the food web. The major decline simulated for the marine birds, observed also in other studies (Bicknell et al., 2013; Heath et al., 2014; Fondo et al., 2015), in the present simulations was caused primarily by the reduction in the prey of Marine birds, since their diet is mainly based on fish as opposed to discards. In particular, this result was anticipated from the mixed trophic impact analysis (MTI, Figure 11) of the mass-balanced model, where there is a strong positive influence of the Bentivorous fish on the Marine birds. Furthermore, Gastropoda presents evidence of indirect effect where the reduced food input as discards is compensated by a decreased predation by echinoderms (Table 16). A similar shift in predation pressure toward prey that remain available after the LO has also been modelled in other ecosystems (Kopp et al., 2016). Mixed responses are present in other functional groups less directly related to discards. For example, the Invertebrate feeding fish benefited from the reduced predation. In spite of possible changes of biomasses of functional groups at sea with potential imbalances and changes in functional role, the evident effect of the LO is the overall small, but still significant, reduction of total ecosystem biomass, including groups mainly comprising species of commercial interest (in NEAS case, Demersal piscivorous fish, Cephalopoda, Flatfish). Warnings about such negative impact of LO were previously raised (Sardà et al., 2015; Bellido et al., 2017) and have been quantified herein, including indirect effects.

The portion of target species smaller than the MCRS that will be landed, has negative repercussions on many other commercially exploited species and, consequently, landings of marketable species are reduced in all fishing gears except mid-water pelagic trawlers and purse seines (Table 16, Figure 45). The reduction of revenues from marketable landings is partially due to the reduced landings of commercial species of higher price. Selling the landed discards (C_{LO}) for industrial processing would represent additional revenue (R_{LO}) that only compensates the decreased revenues from marketable landings. Only for bottom otter-board trawlers the revenues from selling C_{LO} will result in a minimal benefit, while for other fishing activities these revenues would just cover the lost revenues at best. Another big concern highlighted in the present case study is the disproportionality between the increase of workload for landing the discards and the economic benefit derived from it. This in fact could pose a huge obstacle for fishermen's compliance to the regulation (Veiga et al., 2016). The landing obligation, in fact, causes a significant increase of the total landings for all fishing activities, except for the rapido trawl and the dredges. In particular, the otter-board trawlers have the largest quantities of discards and the biggest increase to their landings. This may pose a practical problem on-board of fishing vessels, since LO will require additional storage space (Veiga et al., 2016) as well as increasing the handling and sorting time. LO would slow down the on-board processes and increase the handling costs if more crew is required (Garćia-Rivera et al., 2015). These factors if included are going to further increase the negative balance of the LO application.

Moreover, the use of discarded species not for direct human consumption and the consequent potential economic benefits (see Villasante et al., 2016) needs to be carefully evaluated. For example, discards of small pelagics and horse mackerel could be used as forage fish in Bluefin tuna farming (approximately 10-15 kg of food are required for small tuna to increase 1 kg; Ottolenghi, 2008). Therefore, although the use of discards might have positive effects on the small pelagic species by reducing their harvesting for baitfish, the economic sustainability of such practices (selecting, storing, landing and transporting the discard to fish farms) is doubtful. Notably, even if new economic benefits could be obtained from landing discards, the negative ecological effects of this further uptake of marine resources cannot be avoided. This issue would need to be considered in analyses on economic mitigation to the labour burden induced by LO.

Even if the spatial coverage of the modelled area is limited in comparison with the area of application of the EU policy, the case study is representative of the Mediterranean multi-species multi-gear fisheries and the ecosystem complexity (Garćia-Rivera et al., 2015). The modelling procedure itself does not change the initial ecosystem structure. An alternative approach was adopted by Fondo et al. (2015) by using two mass-balanced models to evaluate the temporal effects of the discard banning in Moreton Bay, Queensland, Australia. Their results also indicate negative impacts on the scavengers. Additionally, the removal of the discards adds additional stress to the ecosystem, indicated by reduction of the cycling index.

Clearly, avoiding fisheries discards should be the priority for preventing an additional mortality to marine populations and this should be a policy in any fisheries worldwide (FAO, 2016). The overall result of the landing obligation policy is a reduction in the ecosystem’s total biomass. Finally, it can be concluded that the application of the Landing Obligation in the Mediterranean without any supporting management plan has negative impacts both on the ecosystem and the fisheries and therefore better solutions to solving the discards problem should be sought.

8. *Study on alternative management scenarios*

This chapter will explore different possible fisheries management scenarios and compare their outcomes and evaluate their suitability and effects on the ecosystem.

In the previous chapter the consequences of the Landing Obligation regulation were explored, and most of them were negative. Therefore, in order to mitigate those negative effects different alternative scenarios will be evaluated and compared. These scenarios are complementary to the already applied LO and represent the approaches the fishermen might choose to tackle discard-related issues and reduce the unwanted effects of the LO regulation. Other than effort control and quotas, the improvement of gear selectivity is viewed as one possible measure to mitigate the bycatch issue within the LO regulation (Batsleer et al., 2016; Campos et al., 2014).

In the last scenario the ecological effects of reaching the fishing mortality reference points for achieving the maximum sustainable yields (F_{MSY}) will be evaluated. The analysis will we focused on the outcomes of applying single species references points singularly and simultaneously.

8.1. Alternative management scenarios

Four scenarios describing alternative LO implementations are defined as: A) introduction of quotas for small pelagic fish; B) halving the fishing effort of otter trawl fisheries; C) improving selectivity of OTB and TBB fisheries; D) application of both quotas for small pelagic fish and improving selectivity for OTB and TBB fisheries. For simplicity’s sake, instead of the NEAS ensemble of models, the current analysis will be carried using the reference Ecosim model described in Chapter 5 (REF) and the same model but with the LO applied as described in Chapter 7 (scenario LO). Alternative scenarios are based on the LO scenario since they represent adaptive measures to LO. The last five years (2026-2030) of the LO scenario and each alternative scenario (A-D) will be compared with the last five years of the scenario without the LO applied (REF).

Scenario A will simulate the introduction of quotas for Small pelagic fish. The situation is equivalent to an inclusion of the discards in an apparent landing quota, assuming a constant effort. From 2020 to 2021 the fishing effort of the PTM and PS was reduced to a level proportional the contribution of the Small pelagic fish marketable catches to their total catches, respectively to 92.2% and 92.1%.

The objective of Scenario B is to reduce the landings subject to the LO by reducing the fishing effort of gears catching the largest quantities of LO subject discards. In the NEAS area most of the discards subject to the LO are caught the bottom otter-board trawl (OTB). Shutting down the OTB fleet is not a socially acceptable solution, therefore the simulation is carried by halving its fishing effort from 2020 to 2021.

In scenario C, a technical measure of improving the selectivity of trawling gear is applied from 2020 to 2021 to reduce the discards. The scenario simulated a change from a 50 mm diamond-mesh to a 40 mm square-mesh codend: effects were represented by modifying landed and discarded quantities of each functional group for OTB and TBB, on the basis of documented selectivity data (Table 18, Eq. 42). The simulated change of selectivity is obtained by multiplying the ratio between selectivity curves and the landed and discarded weight frequency distributions by species, obtained from measurements on-board fishing vessels in the northern Adriatic (Celić, unpublished data), (Figure 46, Eq. 42). Selectivity parameters are evaluated per species when possible, otherwise the whole functional group's selectivity is estimated based on the available species. Each gear's selectivity change ratio is based on the species composing its catches. Since macrobenthic organisms have no possibility of moving actively in search for an escape from the codend and no data were available to estimate their selectivity, a conservative factor (95%) was assigned to Other benthic filter feeders and Annelida & other worms, higher than what is reported by Luchetti et al. (2006) for other benthic functional groups.

Table 18. Selectivity parameters used in the scenario C and D. Abbreviations: no var. – no variations, species names are present in Table 6.

Functional group	Selectivity change ratio				Source
	Marketable catch (C_m)		Discards (OTB, TBB)		
	OTB	TBB	other discard (C_{od})	LO discard (C_{LO})	
Elasm	no var.	no var.	no var.	-	Luchetti et al., 2006; Luchetti, 2008
SmallPel	-	-	49.5%	40.72%	Luchetti et al., 2006; Luchetti, 2008
BenthPel	99.9%	-	78.5%	70.10%	Ozdemir et al., 2012; Sala et al., 2008
CoastPla	-	-	90.0%	-	Luchetti et al., 2006; Luchetti, 2008
PelPisc	no var.	-	-	-	
InvFeed	no var.	-	-	87.27%	Belcari et al., 2011
Detritiv	99.8%	-	98.0%	-	Sala et al., 2008
Herbiv	-	-	-	-	
Flatfish	no var.	no var.	90.0%	-	Guijarro and Massuti, 2006; Sala et al., 2008
Benthiv	90.3%	-	52.0%	74.34%	Deval et al., 2007; Sala et al., 2008; Sala et al., 2015
DemPisc	no var.	no var.	no var.	-	Sala et al., 2010
Cephal	94.2%	97.1%	78.8%	-	Tosunoglu et al., 2009; Belcari et al., 2011
Bivalv	no var.	97.6%	91.8%	-	Luchetti et al., 2006; Luchetti, 2008
AnnWorm	-	-	95.0%	-	A conservative estimate/assumption
DecaSto	96.8%	95.4%	71.9%	-	Deval et al., 2009; Tosunoglu et al., 2009
Gastrop	100.0%	100.0%	91.8%	-	Luchetti et al., 2006; Luchetti, 2008
Echinod	-	-	91.8%	-	Luchetti et al., 2006; Luchetti, 2008
OthBenth	-	-	95.0%	-	A conservative estimate/assumption
MacroZoo	-	-	no var.	-	No variation, since the majority are large jellyfish

$$sel_{ratio_{sp,sort}} = \frac{\sum_{lt=1}^{max}\left[\left(\frac{sel_{SM_{sp,lt}}}{sel_{DM_{sp,lt}}}\right) * \%N_{sp,lt,sort} * W_{sp,lt}\right]}{\sum_{lt=1}^{max}\left[\%N_{sp,lt,sort} * W_{sp,lt}\right]} \qquad \text{Eq. 42}$$

where:

$$sel_{XM_{sp,lt}} = \frac{e^{(v_1+v_2*lt)}}{1+e^{(v_1+v_2*lt)}}$$

$$W_{sp,lt} = a_{sp} * lt^{b_{sp}}$$

$\%N_{sp,lt,sort}$ from length frequency distributions (LFD)

The selectivity ratio $sel_{ratio_{sp,sort}}$ by species (*sp*) and sorting destiny (*sort*, either discard or landed) between the square-mesh and the diamond-mesh is given by the ratio between the sum of estimated total weight of specimens caught with the square-mesh and the total weight of specimens caught with the diamond-mesh. For square-mesh, each weight at length (*lt*) is corrected by the ratio between the square-mesh ($sel_{SM_{sp,lt}}$) and diamond-mesh ($sel_{DM_{sp,lt}}$) selectivity values at that length. The total catch of the square-mesh codend is estimated by summing the corrected weights. The logistic selectivity curve for a certain specie (*sp*) is defined by the parameters v_1 and v_2. The weight of a specie (*sp*) at length (*lt*) is defined by the length-weight relationship parameters *a* and *b*.

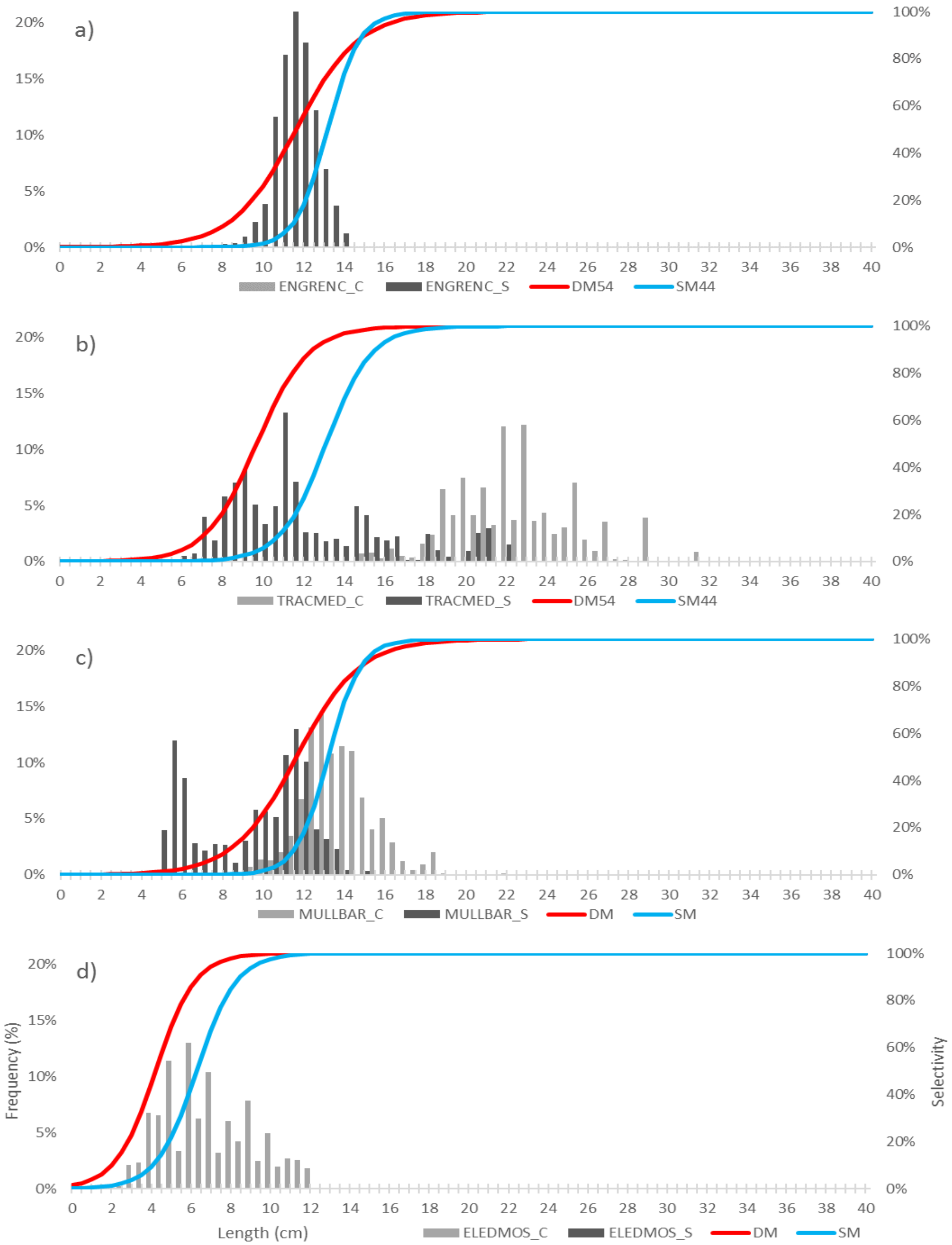

Figure 46. Some examples of gear selectivity modification for some present in NEAS. Plots: a) *Engraulis encrasicolus* (Small pelagic fish), b) *Trachurus mediterraneus* (Benthopelagic fish), c) *Mullus barbatus* (Benthivorous fish) and d) *Eledone moscata* (Cephalopoda). Length frequency distribution of the species is represented by the bars: discard in dark grey, marketable landings in light grey, fish – total length, cephalopoda – mantle length. Selectivity curves are represented by lines: DM – diamond-mesh in red, SM – square-mesh in blue. The length frequency distributions are converted in weight distributions by multiplying for specimens mean weight at length.

In the last scenario D, both the scenarios A and C are applied at the same time to evaluate the improvement of the selectivity for the bottom trawlers and the landing quotas for the PTM and PS.

8.1.2. The effects of alternative management scenario on the ecosystem and the fishery

In order to adapt to the introduction of the LO, all the scenarios involved a reduction of *total* catches with respect to the reference scenario. The application of quotas (scenario A) for mid-water pelagic trawlers (PTM) and the purse seines (PS) resulted in a reduction of their fishing effort by -7.7% and -7.8%, respectively, and a decrease in catches by -7.3% and -7.4%. In scenario B, halving the fishing effort of OTB resulted in decreasing its catches by -46.7%. The modifications of trawling gears' selectivity in scenario C affected primarily the catches of OTB (-15.3%) and to a lesser degree the TBB (-6.6%). The combined application of quotas to small pelagics and increased selectivity for trawlers (scenario D) resulted in decrease in catches for OTB (-15.2%), TBB (-6.6%), PTM (-6.3%) and PS (-6.4%).

Table 19. Summarized effects of the reference LO model and the alternative management scenarios on biomasses and catches compared with the reference model without the landing obligation. Abbreviations: Biom. - total biomass, C_m+C_{od} – part of the catches not subject to LO, C_{LO} - part of the catches subject to LO. The highest values of the four scenarios are in red, while the lowest negative values are in blue. Functional group names as in Table 6.

	LO – Representative LO model			Scenario A - Small pelagic fish quota system (PTM, PS)			Scenario B – trawling effort control policy (OTB)			Scenario C - Trawling gears' selectivity improved (OTB, TBB)			Scenario D - Trawling gears' selectivity improved (OTB, TBB) + small pelagic fish quota system (PTM, PS)		
	Biom.	Catches		Biom.	Catches		Biom.	Catches		Biom.	Catches		Biom.	Catches	
Func.Gr.	tot.	C_m+C_{od}	C_{LO}	tot.	C_m+C_{od}	C_{LO}	tot.	C_m+C_{od}	C_{LO}	tot.	C_m+C_{od}	C_{LO}	tot.	C_m+C_{od}	C_{LO}
MarMamm	-0.18			0.00			8.16			2.88			3.05		
PelPisc	0.08	0.08		0.87	0.37		0.46	0.46		0.88	0.88		1.67	1.16	
MarBird	-1.54			-1.51			7.63			2.61			2.67		
DemPisc	-1.78	-1.78		-2.58	-2.91		2.93	-13.66		-1.33	-1.32		-2.14	-2.46	
MarTurt	0.00			0.07			0.96			0.37			0.44		
BentPel	-0.53	-0.45	-0.45	-0.16	-1.14	-0.10	19.01	-19.93	-35.63	7.35	-3.64	-22.14	7.75	-4.43	-21.87
Elasm	-0.14	-0.14		0.25	-0.19		0.17	-9.39		-2.14	-2.14		-1.75	-2.19	
Cephal	-0.88	-0.88		-0.58	-0.63		1.19	-14.91		-0.25	-3.18		0.06	-2.93	
Flatfish	-0.23	-0.23	-0.23	-0.39	-0.39	-0.39	0.22	-0.35	0.22	0.06	-0.08	0.06	-0.10	-0.24	-0.10
Benthiv	-0.63	-0.63	-0.63	-0.76	-1.15	-0.76	-0.44	-16.28	-50.16	-0.37	-8.17	-25.89	-0.49	-8.69	-26.00
SmallPel	-0.01	-0.01	-0.01	0.46	-7.09	-2.39	0.43	-0.77	-31.53	0.49	-0.47	-37.42	0.96	-7.58	-39.99
CoastPla	-0.02	-0.02		-0.10	-0.61		0.92	-37.38		-0.10	-5.29		-0.18	-5.87	
InvFeed	-0.12	-0.12	-0.12	-0.27	-0.33	-0.27	6.32	1.81	-1.42	1.06	1.06	-0.81	0.91	0.85	-0.96
MacroZoo	-0.03	-0.03		-0.03	-0.03		-0.08	-50.05		-0.08	-0.08		-0.08	-0.08	
MicroZoo	-0.04			-0.02			-0.13			-0.05			-0.04		
Gastrop	-0.49	-0.49		-0.48	-0.48		-0.83	-6.90		-0.28	-1.69		-0.26	-1.68	
DecaSto	-0.25	-0.25		-0.37	-0.37		-1.29	-6.34		-0.50	-5.54		-0.62	-5.65	
Detritiv	-0.01	-0.01		1.19	-0.39		1.15	-1.35		-0.14	-0.15		1.06	-0.53	
OthBenth	-0.08	-0.08		-0.05	-0.05		0.99	-31.21		0.08	-4.93		0.10	-4.90	
MesoZoo	-0.04			-0.12			0.03			-0.14			-0.23		
Herbiv	0.00	0.00		-0.11	-4.53		-0.41	-0.41		-0.03	-0.03		-0.13	-4.55	
Echinod	-0.60	-0.61		-0.55	-0.55		4.68	-38.96		0.40	-7.79		0.46	-7.74	
AnnWorm	-0.05	-0.05		-0.09	-0.09		-0.17	-19.84		-0.08	-5.00		-0.13	-5.04	
Bivalv	0.04	0.04		0.02	0.02		0.03	0.03		0.35	-1.75		0.34	-1.77	
Supraben	0.00			0.01			-0.05			-0.02			-0.01		
Bactpla	-0.04			-0.03			-0.13			-0.06			-0.04		
Phy1Dino	0.01			0.02			0.04			-0.01			0.00		
Phy2Diat	0.00			0.00			0.00			-0.04			-0.03		
AlgSeagr	0.04			-0.07			-0.57			-0.04			-0.16		
POM	-0.03			-0.02			-0.12			-0.10			-0.09		
FishDisc	-9.83			-9.86			-17.57			-13.75			-13.78		
BottDetr	0.00			-0.01			-0.01			0.01			0.01		

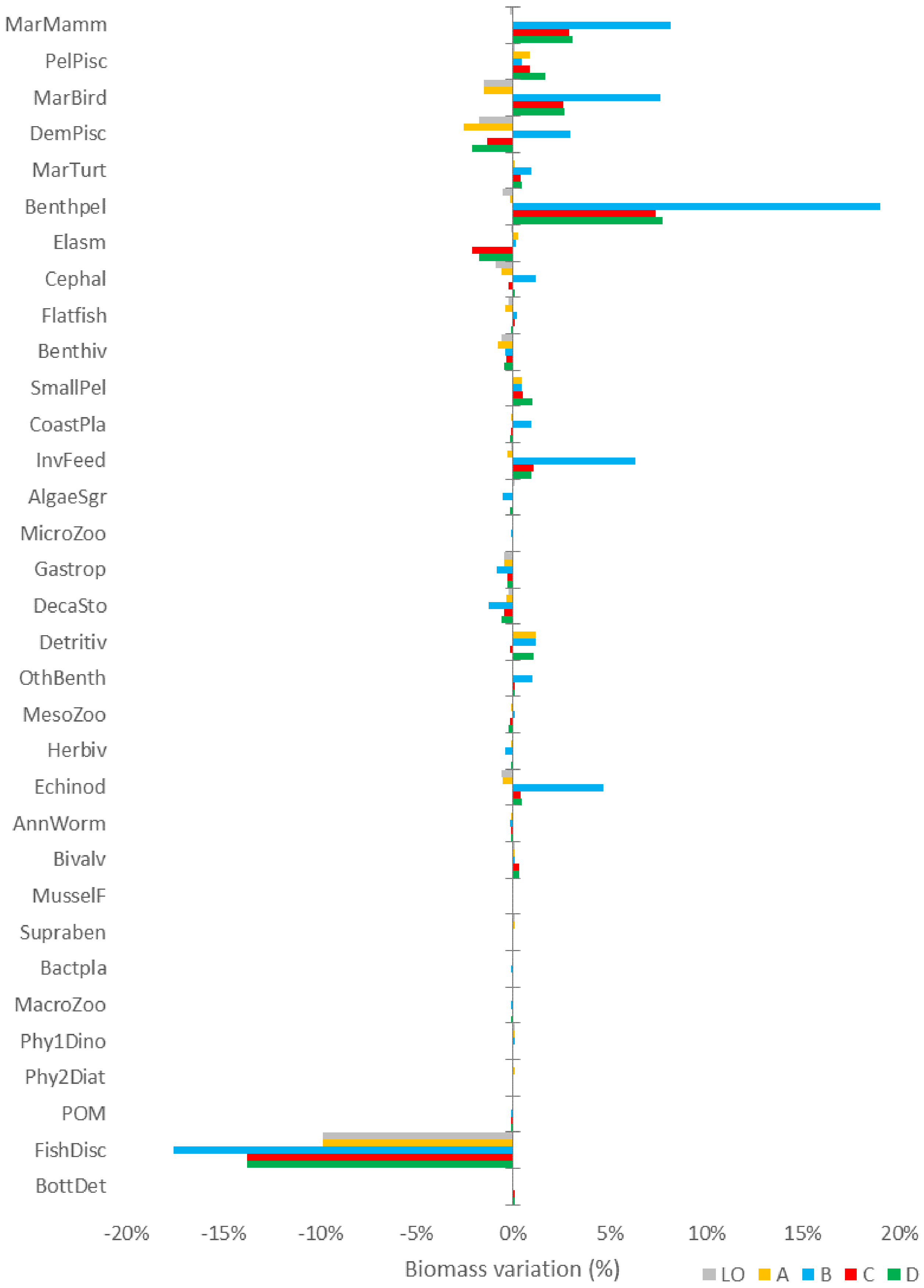

Figure 47. Variation of functional group biomass in the different scenarios compared with the scenario without the LO (REF). Landing Obligation scenario (LO), other scenarios (A-D). Functional group names as in Table 6.

The effects on biomasses and catches by functional group are available in the Table 19 and Figure 47. The largest impacts on the food web were given by scenarios B, C and D. Fishery discards had the biggest reduction in scenario B, followed by scenarios C and D. Marine mammals, Marine birds and Benthopelagic fish benefited the most in scenarios B, C and D, while Demersal piscivorous fish, Invertebrate feeding fish and Echinodermata benefited from the bottom trawlers' effort reduction (scenario B). On the other hand, Elasmobranchii and Demersal piscivorous fish suffered the most negative effects in scenarios C and D. In all scenarios, the planktonic loop was grazed more intensely as result of increased predator populations. Similarly, in scenarios C and D lower biomasses are observed also for Gastropoda, Decapoda & Stomatopoda, Annelida & other worms, Herbivorous fish and the Benthivorous fish.

Moreover, scenario A produced positive effects on the Small pelagic fish population. Negative effects (reduction by less than 3%) were observed for Marine birds, Demersal piscivorous fish, Cephalopoda, Flatfish, Benthivorous fish, Invertebrate feeding fish and Echinodermata, some of them with even lower biomasses than the LO scenario.

Halving the fishing effort of bottom otter-board trawlers produced the greatest increases in biomass and resulted the most favourable scenario for the majority of functional groups. Although the reductions were less than 1.5%, decreases were observed for Gastropoda, Decapoda & Stomatopoda, Herbivorous fish and Macroalgae & seagrass.

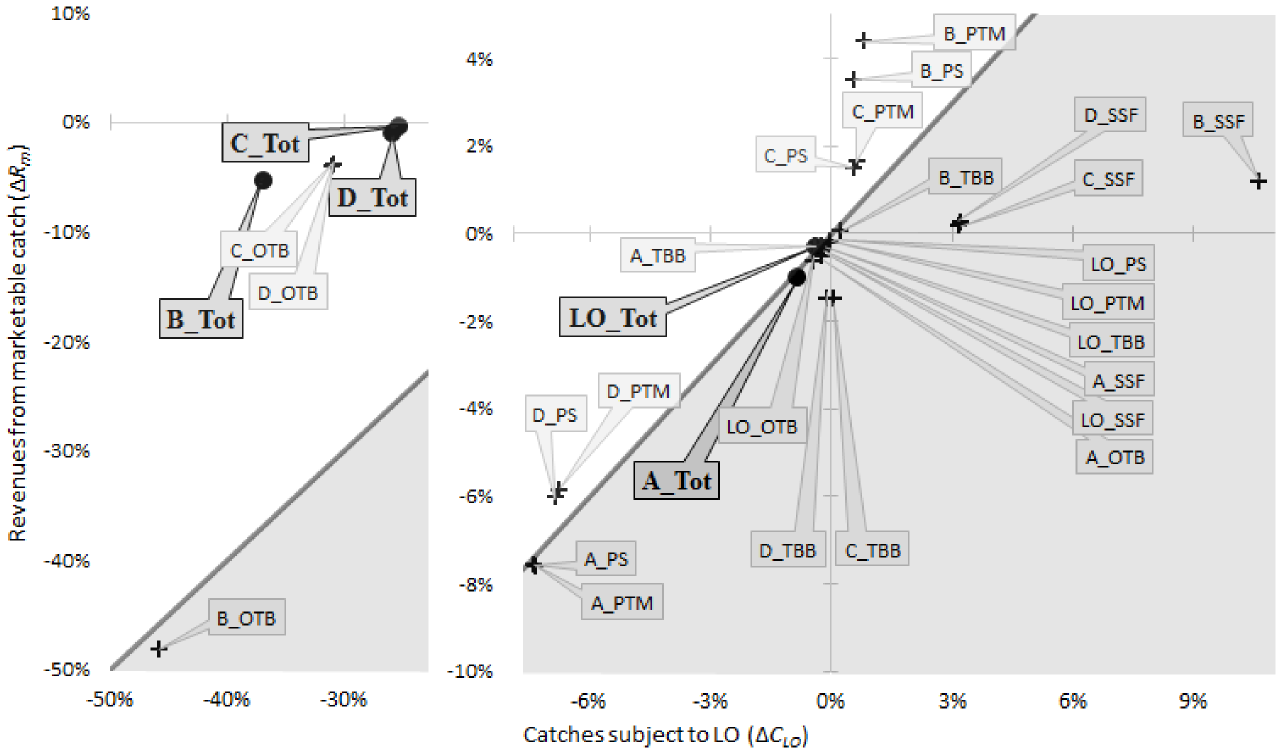

Figure 48. Results of alternative management scenarios with respect to reference scenario (REF), in terms of changes in revenues from marketable catches (ΔR_m) and changes in catches subjected to Landing Obligation (ΔC_{LO}). Changes for each fishing gear (OTB, TBB, PTM, PS, SSF and DRB) and for all fishing fleets (Tot, in bold) are reported for all scenarios (LO, A, B, C, D). The area above the identity line (non-shaded area) indicates the favourable outcomes (changes in revenues are larger than changes on the discards).

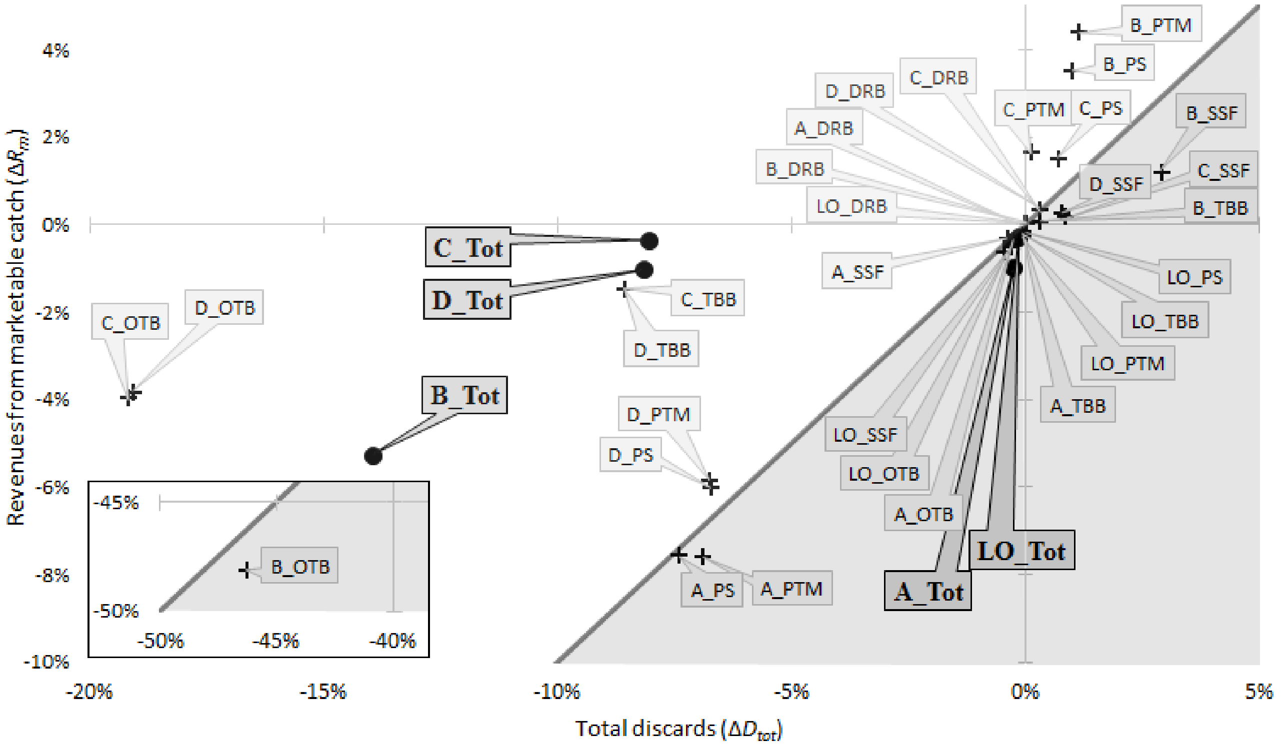

Figure 49. Results of alternative management scenarios with respect to reference scenario (REF), in terms of changes in revenues from marketable catches (ΔR_m) and changes in total discards (ΔD_{Tot}). Changes for each fishing gear (OTB, TBB, PTM, PS, SSF and DRB) and for all fishing fleets (Tot, in bold) are reported for all scenarios (LO, A, B, C, D). The area above the identity line (non-shaded area) indicates the favourable outcomes (changes in revenues are larger than changes on the discards).

Figures 48 and 49 summarise the fishery's and the individual fishing gears' trade-offs between variations of discards (respectively LO subject discards (ΔC_{LO}) and total discards (ΔD_{tot})) and marketable revenues (ΔR_m) as a result of the alternative management scenarios. The most desirable solutions reduce the discards and increase the revenues (4th quadrant), while other acceptable solution that deal with the discards issue either reduce revenues less proportional than the discards (3rd quadrant's upper triangle) or increase revenues more proportional than the discards (1st quadrant's upper triangle). Therefore, intersections of ΔR_m and ΔC_{LO} (or ΔD_{tot}) above the identity line/1:1 line are deemed compliant with the discard issues' policy objectives, i.e. a compromise where the increase of revenues is larger than the increase of discards.

Considering all fleets together, the introduction of the LO (LO_{tot} in Figure 48) resulted in the overall reduction both LO subjected discard (ΔC_{LO} -0.1%) and marketable revenues (ΔR_m -0.3%). Scenario A (A_{tot} in Figure 6) caused a reduction in LO discards (-0.9%), but also a decrease in marketable revenues (-1.0%), resulting in a worsening of the situation with respect to reference conditions (grey shaded area in Figure 48). Scenarios C (ΔC_{LO} -25.4%, ΔR_m -0.4%) and D (ΔC_{LO} -25.9%, ΔR_m -1.1%) showed the best performance with the reduction of LO discards much larger than the reduction of marketable revenues, followed by the scenario B (ΔC_{LO} -37.0%, ΔR_m -5.3%). Results by gear show that scenarios B, C and D have positive effects on PTM and PS, both slightly increasing C_{LO} but less than proportional to the increase in revenues (see B_{PTM}, B_{PS}, C_{PTM}, C_{PS}, D_{PTM} and D_{PS} in Figure 48). Similarly, OTB performed better under scenarios C and D (see C_{OTB} and D_{OTB} in Figure 48).

These results also hold true when considering estimated variations of revenues from marketable catch (ΔR_m) vs. variations in total discards [$\Delta D_{tot}=\Delta(C_{od}+C_{LO})$] (Figure 49). A minor improvement of the fleets' total discard reduction was observed for the implementation of LO and in scenario A (LOtot -0.1%, Atot -0.2%) (shaded area in Figure 49). The total discard reduction was more pronounced in the scenarios C, D and B ($\Delta Dtot$ -8.1%, -8.2% and -14.0%, respectively). The selectivity improvement in scenarios C and D achieved the highest total discards reduction both for OTB (C -19.2%, D -19.1%) and TBB (C and D -8.6%) while marketable revenues reduction was low (OTB: C -4.0%, D -3.8%; TBB C and D -1.5%). This effect is the result of the selectivity improvement in OTB and TBB, which allowed to reduce LO subject catches (C_{LO}) by -31.0% and 0.1% and other discard catches (Cod) by -13.0% and -8.6% respectively. Notably, the mid-water pelagic trawlers (PTM) and the purse seines (PS) benefited from trawlers' improved selectivity.

8.2. Implementation of target fishing mortalities (MSY)

The Maximum Sustainable Yield (MSY) is still estimated in single species stock assessments in the Europe as a reference point indicating when the fishing is considered sustainable (GFCM, 2016b; GFCM, 2016c). The EU regulation EU 1392/2014 also states that by the year 2020 the fishing mortality of European marine resources should be brought to target reference points of MSY.

Therefore, the present analysis aims to explore the implications on the ecosystem of reaching the specie's target MSY. The present simulation is carried by gradually lowering the current fishing mortalities (F_{cur}) to the target fishing mortalities (F_{MSY}). Five economically important Adriatic species, for which single species stock assessments have been performed, have been selected: red mullet (*Mullus barbatus*), common sole (*Solea solea*), mantis shrimp (*Squilla mantis*), European pilchard (*Sardina pilchardus*) and European anchovy (*Engraulis encrasicolus*). The reference values (Table 20), current fishing mortalities (F_{cur}) and the target fishing mortalities (F_{MSY}) are taken from the Scientific, Technical and Economic Committee for Fisheries' stock assessment in 2015 (STECF, 2016) for the *Mullus barbatus*, *Solea solea* and the *Squilla mantis*, and from FAO's General Fisheries Commission for the Mediterranean's stock assessment in 2014 (GFCM, 2016b) for the *Sardina pilchardus* the *Engraulis encrasicolus*. The whole analysis is carried within the MANTIS project, which preferred using STECF reference data for assessments. The present analysis is done using the reference Ecosim model without the Landing Obligation. The target fishing mortalities (F_{MSY}) are applied gradually between 2015 and 2019. The NEAS model does not explicitly represent these species as single species functional groups and, compared to North and Central Adriatic (GSA 17), its area is limited. The functional groups contain a both commercial and non-commercial species and some of them may have even different MSY reference points and different fishing mortalities, as in Small pelagic fish. A compromise is found to apply the single specie F_{MSY} to the functional group proportionally to the specie's weight contribution to the group. Additionally, in the NEAS model the target fishing mortalities (F_{MSY}) values were not applied as they are, but rather the MSY approach was adapted to local fishing mortalities (F_{NEAS}). The fishing mortality in Ecosim is an available forcing function, that when modified to be used in simulations overruns the fishing effort forcing function, and is estimated by default for each time-step by dividing the functional group's predicted catches by the predicted biomass. The NEAS fishing mortality (F_{NEAS}) for each functional group is derived from the Ecosim reference simulation and is reduced proportionally to target specie's (B_{sp}) contribution to functional group's biomass (B_{fg}) by the relative mortality reduction of the functional group (Eq. 43).

$$F_{@MSY_{fg,y}} = F_{NEAS_{fg,y}} \cdot \left[1 - \frac{B_{sp}}{B_{fg}} \cdot \left(1 - \frac{F_{MSY}}{F_{cur}}\right)\right] \qquad \text{Eq. 43}$$

Where $F_{@MSY_{fg,y}}$ is the functional group's fishing mortality in NEAS model at MSY, derived by reducing the reference functional group's fishing mortality $F_{NEAS_{fg,y}}$. Six simulations are carried in total; in five of them the fishing mortality at MSY (F_{MSY}) for each target specie is applied individually, and in an additional simulation fishing mortalities (F_{MSY}) of all target species are applied simultaneously. Other functional groups' fishing mortality or fishing efforts were not modified during these simulations.

Biomass changes were compared across the simulations and an index that assesses whether a fishery is balanced in ecological terms (Fish in Balance index, FiB, Cury et al., 2005) was computed. An increase of the FiB index indicates that the underlying fishery is expanding beyond its traditional fishing area and a decrease indicates a geographic contraction, or a collapse of the underlying food web. The index maintains a value of zero when a decrease in TL is matched by an appropriate catch increase, and deviates from zero otherwise.

Table 20. Details about stock assessment reference points, species contributions to functional group biomass and NEAS original and target fishing mortality. The initial contribution of fishing mortality to functional group's total mortality serves to indicate approximately how much the measure will influence that functional group.

Specie	Source	Year	F_{cur}	F_{MSY}	Functional group	B_{sp}/B_{fg}	F_{Ecosim}	F_ratio	$F_{@MSY}$	Initial F/Z
Mullus barbatus	STECF (EU)	2015	0.540	0.410	Benthivorous fish	1.0%	0.111	99.8%	0.111	7.2%
Solea solea	STECF (EU)	2015	0.620	0.260	Flatfish	22.5%	0.197	86.9%	0.171	18.2%
Squilla mantis	STECF (EU)	2015	0.630	0.560	Decapoda & Stomatopoda	35.6%	0.139	96.0%	0.134	4.6%
Sardina pilchardus	GFCM (FAO)	2014	1.087	0.715	Small pelagic fish	12.2%	0.132	95.8%	0.127	19.7%
Engraulis encrasicolus	GFCM (FAO)	2014	0.990	0.554	Small pelagic fish	81.6%	0.132	64.0%	0.085	19.7%
S. pilchardus					Small pelagic fish	93.9%	0.132	59.9%	0.079	19.7%

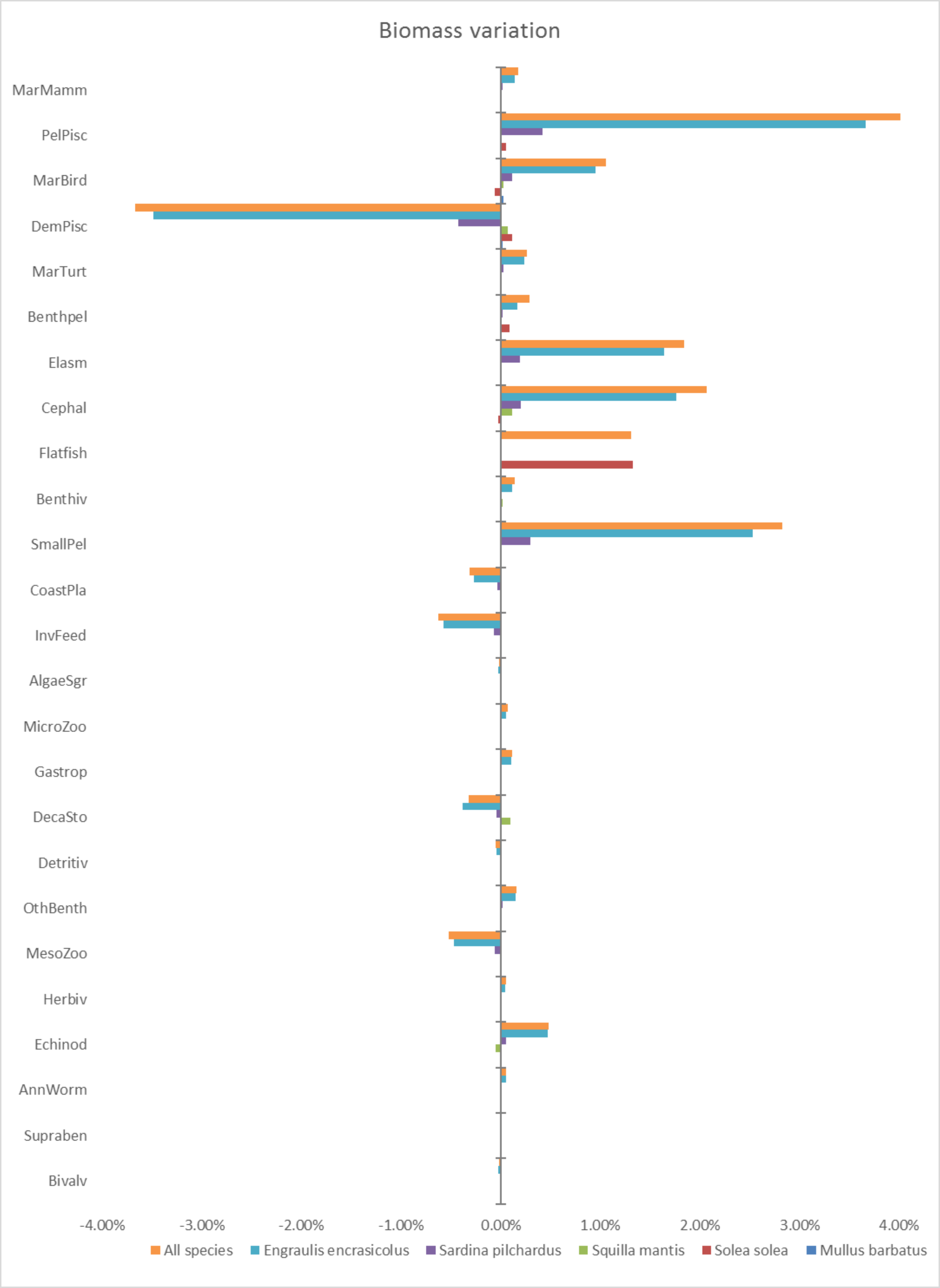

Figure 50. Variation in the biomass of each functional group at different species target fishing mortalities (F_{MSY}). Functional group names as in Table 6.

The effects of the application of target fishing mortalities on the change in biomass of each functional group are shown in Figure 50. The application of the F_{MSY} for *Mullus barbatus* produced the smallest biomass changes, less than 0.03%. The F_{MSY} for *Solea solea* increased the Flatfish biomass by 1.33%, while in case of *Squilla mantis* the biomass of the Decapoda & Stomatopoda only increased by 0.10%, however, both simulations left the changes in biomass of other functional groups smaller than 0.11%. Moreover, in the case of $\mathrm{F_{MSY}}$ of *Sardina pilchardus* major biomass increases were observed in Pelagic piscivorous fish and Small pelagic fish, while Demersal piscivorous fish decreased the most. When $\mathrm{F_{MSY}}$ for anchovy was applied, Pelagic piscivorous fish increased by 3.66%. However, the largest biomass variations were observed when $\mathrm{F_{MSY}}$ was applied for all species simultaneously; Pelagic piscivorous fish increased by 4.13%, followed by Small pelagic fish, Cephalopoda, Elasmobranchii and Marine birds, while Demersal piscivorous fish exhibited the lowest biomasses.

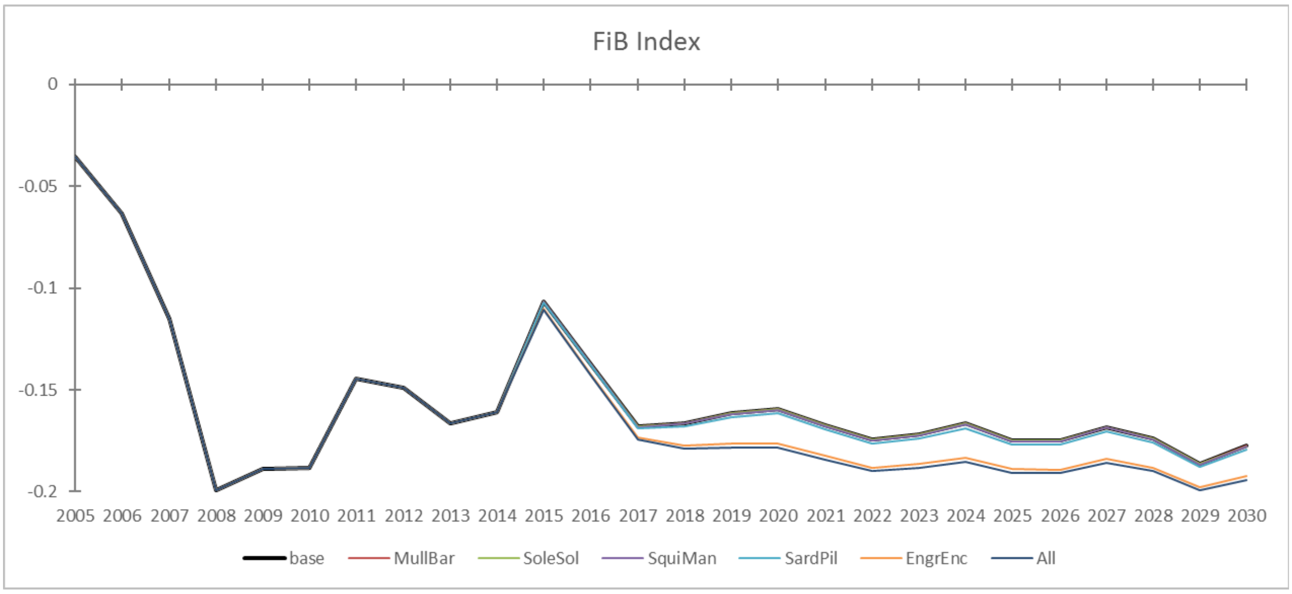

Figure 51. Variation of FiB index across the different scenarios of application of the fishing mortality at MSY. Codes: base – the baseline trend of the model N°4, MullBar – *Mullus barbatus*, SoleSol – *Solea solea*, SquiMan – *Squilla mantis*, SardPil – *Sardina pilchardus*, EngrEnc – *Engraulis encrasicolus*, All – all the species.

Compared to the baseline, the trend of the Fish in Balance index (FiB) showed major decrease only when respectively the F_{MSY} for the anchovy and F_{MSY} for all species simultaneously were applied (Figure 51). In the mentioned two cases the decrease of the FiB index was caused by the decrease of catches (primarily of the anchovy) since the fishing fleet did not expand and the mean trophic level of the catches changed very little.

8.3. Discussion

Alternative management scenarios allowed to assess the effects of catch/effort controls and gear technical modifications on fleet catches, landings, discards and revenues with respect to the LO application. Adaptive scenarios had different effects on the fishing gears but in general terms OTB, TBB, PTM and PS responded to alternative measures, SSF showed some minor variations and DRB resulted almost unaffected (Figure 48 and 49). In scenarios A and B, the effort contraction for OTB or the limitation of catches for PS and PTM reduced discards. However, at the same time revenues were also reduced. For these scenarios to work a change in fishermen behaviour should be done, to effectively improve the selectivity by reducing the discards more than the revenues (Eliasen, 2014; Eliasen and Bichel, 2016). Using more selective gears, as considered in scenarios C and D, is instead a solution that might allow to maintain current fishing capacity and effort and might need less structural interventions on the fishing fleets. Both scenarios C and D exhibit the positive property of reducing the discards more than the marketable part of the catches and may help rebuilding the exploited stocks, as also found in other studies (Coll et al., 2008). Indeed, more selective fishing gears increase escapement of the juveniles and other small-sized species, thus reducing the exerted fishing mortality and reducing the quantities of discards to land. Moreover, our results showed that more selective gears could mitigate, to a certain degree, the loss of marketable catch in the long-term. Also, as a secondary effect, the reduced mortality of the benthic macro-fauna could represent a food source and create the necessary habitat for juveniles and the settlement of larvae, with benefit for exploited stocks. Overall, realistic changes in the selectivity and exploitation pattern (i.e., modification of gear from 50 mm diamond-mesh to 40 mm square-mesh) almost compensated LO impacts on revenues. More importantly, still large quantities of C_{LO} will need to be landed resulting in an inevitable uptake from the system. Therefore, although improving selectivity represents the most adaptive and desirable mitigation measure (Batsleer et al., 2016), in line with the expectations of EU regulation, it will represent an imposed further cost to the sector without solving all issues generated by LO.

The proposed scenarios could be viewed as a top-down approach of fishery management, but bottom-up approaches and proper incentives could also reduce discard (Rochet et al., 2014; Catchpole et al., 2017), even without reducing the profit (Mortensen et al., 2017; Prellezo et al., 2017). Avoiding areas with high bycatch or juveniles by changing the fishing areas, aided by technological solutions as for example fleet information systems, could be an alternative and complementary solution to reduce discards (Eliasen, 2014; Eliasen and Bichel, 2016; Paradinas et al., 2016).

Secondly, the application of the alternative management scenarios had a variety of ecological effects on the ecosystem, contributing to better explain the model’s behaviour. A general trait across the scenarios indicates that the main variations are observed on higher trophic levels predators. In particular, Benthopelagic fish were sensitive to variations in fishing pressure, caused by the high contribution of fishing mortality to their total mortality. Similar patterns are observed for Marine mammals and Marine birds. In Marine birds the effect is caused by high vulnerability values ($v=27$), although the Benthopelagic fish constitute only 1.5% of their diet. The opposite is true for Marine mammals, where Benthopelagic fish constitute the largest portion of the diet while vulnerability is relatively low ($v=2$). This indicates how similar trophic effects and strong trophic bonds can be achieved in different ways. Similar population increases resulted from an improved selectivity simulated in the South Catalan Sea by Coll et al. (2008b). However, although the selectivity pattern was similar, the high contribution of fishing mortality to total mortality caused the biomass of some functional groups to increase between 100% and 600%, values much higher than the results obtained in the NEAS area. The dampened effect in the NEAS model may be caused by: i) a more detailed diet resulting in a better connectivity between functional groups, ii) lower contribution of fishing mortality to total mortality and lack of predation mortality. The analysis gave insights into the importance of trophic interactions on the outcomes of a management scenario. Under single-species approaches the Benthivorous fish should have benefited from reduced fishing mortality by use of more selective fishing gears, reducing their discards and the unwanted catches. However, in the NEAS model, the biomass of Benthivorous fish did not increase because the reduced fishing mortality was compensated by an increase of predation mortality by Marine birds and Benthopelagic fish.

A limitation of the simulated improvement of selectivity is the lack of ability to account for demographic changes in the structure of the population. The escaped resources contribute to the population’s biomass also through their growth, not only as uncaught biomass. However, in the present study no escapement mortality was simulated and its simulation might reduce the positive effects of modifying the gear. Structuring the population by age and using multiple-stanzas in future developments of the NEAS model could contribute to more realistic simulations. Also, additional field studies would allow to compare the effects of improved gear selectivity with simulated data.

The application of a quantitative ecosystem approach to fisheries can give precious insights into long-term ecological and economic effects with reasonable confidence. One of the aims of the EU regulation (EU, 2014) is to reach MSY levels of the exploited stocks by 2020. On the other hand, the implementation of the MSY approach for the different species produced different effects. Large variations were observed only in cases of the anchovy F_{MSY} or MSY applied to all the species together. In these two cases the decreased FiB index indicates a fishery contraction, since the effective fishing effort did not change, while overall decreasing trend was caused by the historic decrease of fishing effort.

Although the application of F_{MSY} for anchovy and management scenario A (quotas for small pelagic fish) have similar objectives (reduction of fishing mortality), the trophic responses and ecological consequences were different. When applying F_{MSY} for anchovy, the populations of Pelagic piscivorous fish increased, while no changes were detected in Benthopelagic fish and Marine birds.

However, the reference points in this analysis were based on single species stock assessments, disregarding the possible effects of trophic interactions. Fishing pressure will be modified to reach those targets and such changes will likely have trophic interactions with other species (Smith et al., 2011). Such modifications may involve fishing effort, spatio-temporal avoidance or modifying fishing gear selectivity. However, the indirect effects of these measures on target species may not be in line with the management objectives (Walters et al., 2005). For example, the reduction of fishing effort by a certain amount to reach the desired target specie's F_{MSY} based on a previous stock assessment may be reached as a management measure, but the desired fishing mortality may not be reached because the target specie's predation mortality increased as a result of trophic interactions with other species. Additionally, reaching simultaneously the target fishing mortality for multiple species could be unlikely because species may share the same fishing gear and different fishing pressure levels would be needed for each specie (Maunder, 2002). A single-species based management may underexploit or overexploit other resources, similar to the effect of "choke species" in the quota system in the Atlantic (Eliasen, 2014).

Therefore, a global MSY estimated through an ecosystem oriented approach may be a suitable reference point for better fisheries management (Maunder 2002). Case studies with multi-species models produced higher estimates of combined yield for aggregate maximum sustainable yield (MSY) targets than single species models, but were more conservative than single-species models when individual MSY targets were used (Holsman et al., 2016; Stäbler et al., 2016). Further, interactions among species require multi-species (MMSY) definitions, incorporating ecological, but also, economic and social considerations (Voss et al., 2014). Multiple stock reference points could be obtained using optimization of bio-economic models (Ulrich et al., 2002; Guillen et al., 2012; García et al. 2016) and estimates such as multi-species maximum economic yield (MMEY) (Voss et al., 2014). However, ecological conditions change over time and target reference points may be sensitive to the changes in natural mortality rates (Collie and Gislason, 2001). For example, the fishing mortality reference levels of prey species should be conditioned on the level of predation mortality.

A limit of the present analysis is the model structure itself, not designed for detailed analysis on single species. In the present model the species are aggregated in functional groups, therefore the effects of reaching MSY for a target specie are not directly observable. A model structured where the target species of a management are represented by a single specie group would allow more effective analysis. Additionally, representing the target specie as a multiple-stanzas would allow to represent and observe the changes in the population's age structure that result from reaching F_{MSY}.

The observed variation of applying MSY fishing mortalities did not produce large variations in the NEAS model, but those variations allowed to investigate the ecologic effects of such measures. Although in the present analysis no economic evaluations were done, good practice would be to evaluate the maximum economic yield as an additional information in evaluating the MSY.

9. Discussion and conclusions

9.1. Summary and discussion

In the present thesis, an ecosystem model of the North East Adriatic Sea was developed, validated, and used as a proxy for Mediterranean mixed fisheries systems and as a potential tool for operational fisheries management.

Input data were gathered and integrated into the ecosystem model and basic properties of the mass-balanced model were assessed. Further, the time-dynamic model was fitted to observed time series data and an analysis to changes in initial parameters and sensitivity to changes in fishing effort was carried out. An ensemble of models with different fitting strategies was created to simulate the uncertainty around trophic interactions and validate the ensemble for further operational use. Possible impacts of the EU Landing Obligation regulation on ecological and socioeconomic aspects of Mediterranean fisheries were assessed, as well as, alternative management scenarios to Landing Obligation and the implementation of MSY reference points were compared.

The NEAS represents a complex eutrophic ecosystem with rich biological communities and a developed multi-gear multi-species fishery. The NEAS model allowed to integrate the available data (from resources at sea to the fishing activities) and represent ecosystem structure. Although with its 33 functional groups, the NEAS model might be considered of intermediate complexity for EwE standards, it allowed describing the dynamics of biological populations and the fisheries trends. The selected level of complexity was also detailed enough to be used for more in-depth comparisons of alternative fisheries management scenarios.

The decisions upon which model structure was based proved to be fundamental and affected the following analyses as confirmed by the Monte Carlo analysis. For example, one of these decisions was the setting of ecotrophic efficiency values close to one. When building the Ecopath model the tendency is to represent the total mortality (predation and fishing) as much as possible, i.e. obtaining ecotrophic efficiency values close to 1 (Christensen et al., 2008). However, although this means the functional group is better connected to the food web, this setting will also limit the changes of initial parameters in Monte Carlo analysis. Indeed, the Monte Carlo analysis allows only parameter combinations that result in a mass-balanced model, therefore if constrained to maintain the model mass-balanced the estimated parameters are likely to be shifted off their initial value. In the NEAS model this was particularly true for Bottom detritus group, where the decision not to export the detritus and to recycle it within the model had a major role. The importance of Bottom detritus in the

NEAS model was evident also through the Lindeman Spine (Figure 14), that showed that most of the trophic flows originated from the detritus. Since Bottom detritus was almost entirely consumed by the food web (99.71%), only the parameter combinations that would increase its quantity were allowed. Additionally, the Monte Carlo analysis also confirmed the results from the keystoness analysis (Figure 12). Functional groups that had major shifts from the initial parameter values (Figure 17) were the groups with highest keystoness index. Primary producers (dinoflagellates excluded) and the planktonic loop (in particular the bacterioplankton), groups closely related to Bottom detritus, had an important role in shaping the structure and dynamics of the NEAS model. Other cases pointed to the cycle between a shared prey, Decapoda & Stomatopoda, and its two predators, Benthivorous fish and Cephalopoda; as well as the top predator, the Elasmobranchii.

A good connectivity of functional groups in the NEAS model is pointed out by the mixed trophic impact analysis and the correlations of corrected biomass and initial parameters variation in the Monte Carlo analysis. The former, identified the possible effects of trophic interactions between biological resources or between biological resources and fisheries (Figure 11), observed also in all further Ecosim simulations. The latter (Figure 22.c), highlighted how, except for groups with weaker trophic interactions (Figure 12), variation of initial parameters do not allow to predict linear biomass changes, instead non-linear trophic interactions took place.

The validity and robustness of NEAS mass-balanced model is confirmed by the ranges of trials, with the five lowest relative SS (Figure 19) being around the NEAS initial parameter values. If the opposite had been true, then those ranges would have been smaller and offset from the initial parameters.

Moreover, sensitivity analysis of the influences of fishing effort forcing functions (Mackinson et al., 2009a) indicated the dynamics of the NEAS model are more influenced by the primary production than by the fishery exploitation (Figure 24). The low impacts of the application of MSY reference points on the biomass of each functional groups (Figure 50) confirm the previous result.

In this thesis, an ensemble of models based on different fitting strategies was proposed as a new method to estimate uncertainties of trophic interactions. Such method is undocumented and represents an unexplored concept, and application, in EwE modelling. The method was based on an expert driven recursive fitting procedure that produced an ensemble of models accounting more realistically for the complexity of the system. Although it was not always successful, this approach allowed to optimize models to better fit the dynamics of certain functional groups (Figure 26, 27, 32 and 36). The created models would have different trophic properties even if their goodness of fit indicator (SS) may have had similar values (Figure 27). The variability obtained between the different simulations is an indicator of uncertainty of the ensemble's trophic properties, a source of uncertainty accounted for when using the ensemble for operational purposes, as in the present evaluation of the Landing Obligation regulation. The obtained trophic responses differ from simulation to simulation (Figure 28 and 29) and also the simulations' variability is larger (Figure 30) than the variability in the Monte Carlo trials (Figure 22.a). The latter usually shifts the predicted values of the simulation by a same amount compared to the reference simulation, especially when the initial biomass is modified (Figure 22.b).

In the NEAS, during the last decade, the biological populations underwent complex dynamics (Figures 3 and 4) and there was a general decreasing fishing effort trend (Figures 5 and 6). Nonetheless, the ensemble of models was able to reproduce the observed biomass and catch trends (Figures 28 and 29). Biomass trends of the macrobenthic fauna were predicted more accurately than highly mobile fish species (Figures 31.a, 33 and 38). Additionally, the different trophic responses of each functional group also induced large variation in functional group accuracy and the correlation between predicted and observed data (Figure 33, 34 and 35). Biomass predictions had lower accuracy, precision and correlation (Figure 31.a, 33 and 38) than catch predictions (Figure 31.b, 34 and 39), a situation similar to the model described by Ortega-Cisneros et al. (2017). In particular, biomass simulations had a tendency to be overestimated and uncorrelated (Figure 31.a, 33) while catches had positively correlated trends with most biases smaller than 50% or within one standard deviation of the observed data (Figure 31.b, 34). Overall, the combined predictions of functional group mean biomass and catch trends were accurate (Figure 40) or slightly overestimated (Figure 31.c), with a good precision and low positive correlation (Figure 40), therefore the NEAS ensemble of models was deemed a reliable operational tool for further fisheries management analysis.

All this allows the exploration of potential ecosystem-level effects of fishery management measures. In particular, the implementation of the EU Landing Obligations in the context of the present case study, was assessed. The analysis showed that LO implementation significantly reduced the input of fishery discards in the food web. Major variations of functional group biomass were observed because of indirect effects through the food web, where Marine birds had the major decline in biomass. The change of ecosystem structure was predicted by the mixed trophic impact analysis (MTI, Figure 11), but those changes also highlighted the role of keystone species (Figure 12), like the Benthivorous fish, an important prey for Marine birds. Moreover, effects of LO on fisheries activities highlighted that the otter-board trawlers would have the largest increase in workload, but without an adequate increase in revenues. The disproportionality between the increase in workload and revenues could pose an obstacle for fisher compliance with the regulation. The analysis highlighted that the Landing Obligation would cause a reduction of ecosystem biomasses and increase fisher workload while reducing fisheries revenues. Such results, although based on the NEAS case study, can be considered as representative of the potential impacts that the EU LO could have on other mixed fisheries without quotas.

Clearly, avoiding fisheries discards should be the priority for preventing an additional mortality on marine populations and this should be a policy in any fisheries worldwide (FAO, 2016). However, since complete avoidance is unrealistic the best option for discards seems that of returning them to sea, where they can be food for marine organisms. By doing so the organic material is recycled back through the food web (see also Heath et al., 2014; Fondo et al., 2015). In contrast, landing discards represents a further removal of energy from the system and promoting their use as raw material for industrial processing is contradictory. This latter strategy could have short-term economic benefits, but it may not be sustainable in the long-term. If the objectives of the regulation were to be achieved and discarding were reduced, shortages of raw material to sustain the processing plants may occur in the future. The observed decreasing trend of the fishmeal stock prices could also reduce the potential economic benefits of selling discards. Without long-term sustainable strategies also accounting for the impacts/achievements of LO, building the processing capacity on the present situation may lead to future overexploitation of the resources and to unnecessary costs.

The assessment of different alternative management scenarios gave precious insights into the options available for tackling the discards problem and into how do multiple policy schemes interact with each other. The effort control policies (scenarios A and B) are not viable solutions for discard reduction because they reduce all the catches proportionally (Figures 48 and 49), instead of putting in place appropriate behaviours to avoid discards (Eliasen, 2014; Eliasen and Bichel, 2016). On the other hand, the improvement of selectivity in scenarios C and D managed to reduce the catch of LO-subjected discards and also had positive effects on other fishing gears (purse seine and mid-water pelagic trawl) (Figures 48 and 49). The use of an ecosystem model allowed an integrated study of the ecological consequences of modifying the catch composition by changing the gear’s selectivity (Fauconnet and Rochet, 2016). Changes in functional group biomasses exposed the effects on the food web (Figure 47), in particular, highlighting the connectivity of top predators (Marine mammals and Marine birds) to the food web and their susceptibility to indirect effects.

The merit of the regulation is to further push the scientific and fishing community to improve selectivity and the adoption of quotas, which are the two pursuable directions to contrast the LO negative effects in the Mediterranean Sea. However, although spatial regulation and temporal restrictions to avoid catching juveniles together with changes in gear selectivity might result in some improvement in the selectivity pattern, discardless fishing remains a utopia. Moreover, it should be considered that by forcing adaptation at the cost of the sector, the regulation might dangerously result in unwanted directions and negative results (e.g., development of black market and no solution of overexploitation). The analysis of adaptation scenarios to LO shed light on the evidence that management of fisheries requires to deal with each system-to-be-governed in its own specific way, whether this will be achieved through a combination of improving the selectivity, effort/catch control, fishing closures or any other appropriate measures (Sardà et al., 2015).

9.2. Limitations and future development

Even if the NEAS model was able to reproduce the observed historical trends, its limited spatial coverage and the open boundaries leave the simulations vulnerable to effects of species' migration. As most of the Adriatic Sea, the NEAS is subject to strong seasonality which drives species' migration; therefore, extending the modelled area to cover the whole Adriatic Sea would reduce the migration effect and make it a factor internal to the model. Further improvement would be achieved by representing the migration effects by spatial simulations in Ecospace model, which would need the availability of spatio-temporal data. The inclusion of environmental dynamics would also improve the present model.

Moreover, the Monte Carlo analysis applied in this thesis allowed to investigate the parameter combinations for alternative mass-balanced models. However, due to its constrains, the analysis could not be considered a full sensitivity analysis to parameter variations. A development of a time-dynamic sensitivity analysis would allow to better investigate model behaviours and the relationship with parameter variations. Instead of using the actual EwE GUI, a solution could be to drive the EwE models from external scripting languages, allowing user defined customization for specific research purposes. The present EwE suite integrates the Stepwise Fitting Procedure for automated fitting and testing of alternative hypotheses (Scott et al., 2016), however the development of automated fitting procedure for creating ensembles of Ecosim models would be beneficial.

When considering the simulations of LO alternative management scenarios, further potential refinement of the approaches applied in this study could be identified. For instance, although the approach for simulating the improvement of selectivity was conservative, the effects estimated in scenarios C and D might be better explored explicitly accounting for ontogenetic and demographic changes in the structure of the population. Thus, further analyses might be performed to properly simulate these effects by including multi-stanza representations (Walters and Martell, 2004). Moreover, in this study, the effects of the application of exemptions rules to LO were not tested. Although these measures will reduce the implementation of the LO and its absolute effects, the direction of changes presented here still hold for a partial LO application.

9.3. Conclusions

The NEAS model enabled to integrate available data, producing a description of the dynamics of the NEAS over the last decade. Moreover, due to the complexity of the ecosystem, the multi-target nature of fisheries exploitation and presence of mixed fisheries, the NEAS model could be considered as representative of the Mediterranean coastal fisheries. The use of an ensemble of calibrated model simulations well represented the dynamics of resources and catches in the past while providing indications on model uncertainty. Moreover, the structural properties of the mass-balanced model have an important role in the following time-dynamic simulations and they constrain the Monte Carlo alternative mass-balanced model solutions. Globally, the results for future scenarios were consistent and can be considered robust, at least in terms of identified directions of change. Top predators, Marine mammals, Marine birds and piscivorous fish, are strongly affected by the indirect effects of management policies Although the effect of LO are site-specific, due to local fisheries and resource conditions, the general conclusion from explored scenarios can be generalized to any area where mixed fisheries are not regulated by quotas. The implementation of the EU LO will have negative consequences by reducing the ecosystems biomasses, increasing workload for fishermen and reducing the fisheries economic revenues. The negative effects of the LO implementation cannot be removed by the adoption of adaptive strategies considered here. Among those, the improvement of gear selectivity is the best adaptive solution to cope with the regulation.

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