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

10–14 October 2016

Reykjavik, Iceland

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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Executive summary

The Working Group on Multispecies Assessment Methods (WGSAM) met in Reykjavik, Iceland, 10–14 October 2016. In this tenth report of the pan-regional WGSAM, work focused on four (B, E, F, G) of the multi-annual ToRs.

Based on their knowledge, participants provided an updated inventory of progress of multispecies models in ICES Ecoregions (ToR A), noting those regions where no information was available. Reporting on ToR A was scarce compared to previous years, partly because recent relevant work was reported against ToR E and G instead.

A Key Run (ToR B) of the Baltic Sea Ecopath with Ecosim (NS-EwE) model was presented and reviewed in detail by 4 WGSAM experts, and approved by the group following implementation of changes agreed in plenary at the meeting and verified by the 4 experts in January. The Key Run is documented in a detail in Annex 3, with key outputs summarised in Section 3 and data files made available on the WGSAM webpage). WGSAM also conducted an informal review of the LeMans modelling framework for potential application in the Irish Sea, and recommended adjustments to the framework for further review. Because the LeMans framework is a within-model ensemble addressing parameter uncertainty, this review also related to ToR D.

Multispecies model skill assessment (ToR C) and multi-model ensemble methods (ToR D) were not emphasized this year. However, plans were made to coordinate future work for ToR C, and one ToR D presentation reviewed the utility of a dynamic multimodel ensemble for making inferences about the real world. This method can infer results for individual components of aggregate groups; the ensemble model uses correlations in other ecosystem models to determine what the models that group species would have predicted for individual species. A proof of concept for the North Sea was presented.

Ecosystem indicator analyses (ToR E) were presented from a wide range of ecosystems. A theoretical analysis comparing results from the Celtic and North Seas with 4 "idealized" fleets was presented to analyse the performance of selected indicators in a multispecies mixed fishery. Four indicators including the Large Fish Indicator (LFI) were examined, and shown to have mixed utility in measuring the impact of different fleet sectors, with the best indicator varying by ecosystem. A multivariate analysis of ecosystem responses to multiple drivers was conducted for four US ecosystems using gradient forest method to identify potential ecosystem thresholds. Other multivariate methods were reviewed that draw on the strengths of multiple indicators for the Northeast US shelf ecosystem. A food web based biodiversity indicator was presented with an application for the Baltic Sea. This could be extended to any ecosystem with an EwE or similar model. A community status indicator relating a species-area relationship to the LFI and mean trophic levels was presented for the Swedish west coast.

Impacts of apex predators on fisheries (ToR F) were examined with one presentation and a group discussion planning further work. A multipecies production model was parameterized to simulate interactions between three fish guilds, fisheries, and one marine mammal guild, concluding that fish reference points and trajectories change with marine mammal interactions. Fishery management was also important to reduce vessel interactions with and ensure prey supply to marine mammals.

Exploration of practical advice for fisheries management incorporating multispecies, mixed fishery, and environmental factors (ToR G) was evident across regions. Two approaches for incorporating species, fleet, environmental, and other interactions are in progress in the Northeast US. One presentation outlined the New England approach, and another outlined the Mid-Atlantic approach. In New England, a management strategy evaluation is in progress to evaluate harvest control rules that consider herring's role as forage in the ecosystem. The modelling framework and stakeholder workshops were discussed. In the Baltic, a Nash Equilibrium optimisation approach incorporating environmental factors was presented for the cod-herring-sprat fishery to attempt to identify a solution that would give good yield for all species simultaneously. In the North Sea a theoretical analysis using 4 “idealized” fleets was presented to analyse the potential implications of "Pretty Good Yield" ranges around MSY. The model examined the likelihood of the fishery being precautionary for the different species given the uncertainties involved, and concluded that the upper ends of MSY ranges would not guarantee precautionarity.

1 Opening of the meeting

The Working Group on Multispecies Assessment Methods (WGSAM) met in Reykjavik, Iceland, 10–14 October 2016. The list of participants and contact details are given in Annex 1. The Terms of Reference for the meeting (see section 1.2) were discussed, and a plan of action was adopted with individuals providing presentations on particular issues and allocated separate tasks to begin work on all ToRs.

1.1 Acknowledgements

WGSAM would like to thank Bjarki Þór Elvarsson, Hoskuldur Bjornsson, and Guðmundur Þórðarson for logistics during the meeting, and Maria Lifentseva of the ICES Secretariat for her continued support with the WGSAM SharePoint site.

1.2 Terms of reference

The Working Group on Multispecies Assessment Methods (WGSAM), chaired by Daniel Howell, Norway, and Sarah Gaichas, USA, met in Reykjavik, Iceland, 10–14 October 2016 to:

Work on all tors. Focus on B, E, F, G (in bold).

ToR A. Review further progress and deliver key updates in multispecies and ecosystem modelling throughout the ICES region

ToR B. Update of key-runs (standardized model runs updated with recent data, producing agreed output and agreed upon by WGSAM participants) of multispecies and ecosystem models for different ICES regions (Baltic Sea EwE, LeMans Framework proposed for use in Irish Sea).

ToR C. Consider methods to assess the skill of multispecies models intended for operational advice.

ToR D. Investigate the performance of multi-model ensemble in comparison to single model approach

ToR E. Test performance and sensitivity of ecosystem indicators

ToR F. Metanalysis of impact of top predators on fish stocks in ICES waters

ToR G. Explore the consequence of multispecies, mixed fisheries interactions and environmental factors in practical multispecies advice for fisheries management (MSY related and other biological reference points)

2 ToR A: Review further progress and deliver key updates in multispecies and ecosystem modelling throughout the ICES region

The review of progress of multispecies models in ICES Ecoregions given below is not intended to be comprehensive and exhaustive. It reflects the knowledge available to the participants at the meeting and input from WGSAM who were not able to attend in person.

There was no participation from Russia or Canada at this year's meeting, and consequently no update on modelling from the regions.

2.1 Ecoregion A: Greenland and Iceland Seas

There is no progress to report on multispecies modelling in this Ecoregion this year.

2.2 Ecoregion B: Barents Sea

Work is progressing on the Atlantis model for the Barents and Norwegian Sea.

The "SYMBIOSES" project aims to combine a multispecies fish model for the Barents Sea with detailed larval and oil modelling in the spawning grounds in the Lofotens. The aim of this tool is to evaluate the potential impacts of oil spills on our main fishing species. The first stage of the project is now over, and the model has been developed and tested for cod, and seems to perform "sensibly" for this stock. Future development, for example other species or development of the ecotoxicology, is dependent on future funding.

The Nordic and Barents Sea Atlatins (NoBa) model was (and is still being) developed to explore combined climate and fisheries scenarios in the Norwegian and Barents Sea (Hansen *et al.*, 2016). Snowcrab has been added as one of 53 components that represent the ecosystems of the area. Fisheries are represented as time-series of fisheries mortalities, and the biomasses and catches are reasonable. NoBa is presently being used to study the snow crab population development in the Barents Sea, changes in horizontal distribution of large commercial stocks, impact of ocean acidification and climate change on lower trophic levels and its impact on the commercial stocks, evaluation of indicators used in the Barents Sea management plans, and will also undergo a skill assessment in early 2017.

There is a new multi-year project "REDUS" aimed at quantifying and reducing uncertainty in our stock assessments http://www.redus.no/. Part of this project will be to produce a MSE tool that can connect to different operating models, including multispecies models. This will allow the performance of the single species assessment model and the HCR against a range of uncertainties in operating model specification. This project is at the stage of hiring staff, and will be presented in more detail in future.

The cod HCR has been in place in the Barents Sea for a number of years and is up for revision. As part of this evaluation, a number of potential new HCRs were evaluated. In addition to standard hockey stick HCRs, one of the HCRs called for increasing F at high stock sizes, while another called for this increase in catch only when the cod stock was high and the biomass of the main prey species (capelin) was forecast to be low. These alternatives have been sent to the Norwegian–Russian fisheries commission, which is due to decide the new HCR by the end of October 2016.

2.3 Ecoregion C: Faroes

There is no progress to report on multispecies modelling in the Ecoregion this year.

2.4 Ecoregion D: Norwegian Sea

Progress here is presented in conjunction with work in the Barents Sea under ecoregion B.

2.5 Ecoregion E: Celtic Seas

2.5.1 Update on Celtic Sea LeMans Model

We have continued to develop a Celtic Sea version of the LeMans model (Hall *et al.*, 2006; Thorpe *et al.* 2015). The main developments have been application of a variable length discretisation, starting at 2cm, and gradually increasing, whilst keeping the number of size classes at 32. The effect of this was increase the ability to resolve the growth trajectories of the smaller stocks such as sprat or boarfish, without significantly increasing model run time, except for the fact that the time-step also had to be decreased from 1/10th to 1/20th of a year. The default life history parameters for the model are shown in Table 1:

Table 1. Central estimate for life history parameters for the 18 Celtic Sea model stocks.

Life History Parameters

STOCK	LINF	LMAT	VBG K	Alpha * 1000	beta
Sprat	17.98	9.96	0.57	2.112	3.4746
Boarfish	21.10	11.60	0.53	221.8	1.9707
Poor Cod	21.10	11.60	0.53	9.2	3.0265
Norway Pout	25.25	13.75	0.49	5.18	3.117
Herring	33.51	17.99	0.43	6.03	3.0904
Blue Whiting	37.63	20.09	0.41	5.18	3.117
Dab	40.72	21.65	0.39	7.4	3.1128
Horse Mackerel	42.77	22.69	0.38	3.4	3.2943
Mackerel	47.90	25.26	0.36	3.01	3.29
Megrim	69.35	35.91	0.31	6.4	2.9931
Sole	69.35	35.91	0.31	3.6	3.3133
Whiting	70.37	36.41	0.30	5.18	3.117
Plaice	71.39	36.91	0.30	21.5	2.7901
Haddock	83.59	42.88	0.28	5.58	3.133
Monkfish B	104.87	53.19	0.25	15.3	2.9979
Hake	121.04	60.95	0.24	4.7	3.099
Cod	125.08	62.88	0.23	6.03	3.0904
Monkfish P	152.28	75.80	0.21	15.3	2.9979

Default / Central Estimate Values :: Uncertainty +/- 20% for LINF

The changes in length discretisation have small impacts on 17 of the stocks, but were necessary to improve the dynamics of the sprat stock. Previously this stock was unstable to even very low levels of fishing (WGSAM, 2015), but now it is stable to fishing at up to 1.5x single species FMSY. The model stock biomass is relatively small at 10–20k tonnes, but this level of sprat is believed to be reasonable for the Celtic Sea region. Given the changes in the behaviour of the sprat stock to this modification of numerics, it is important to test the model response to even finer resolution of length classes, and it is planned to do this in the future.

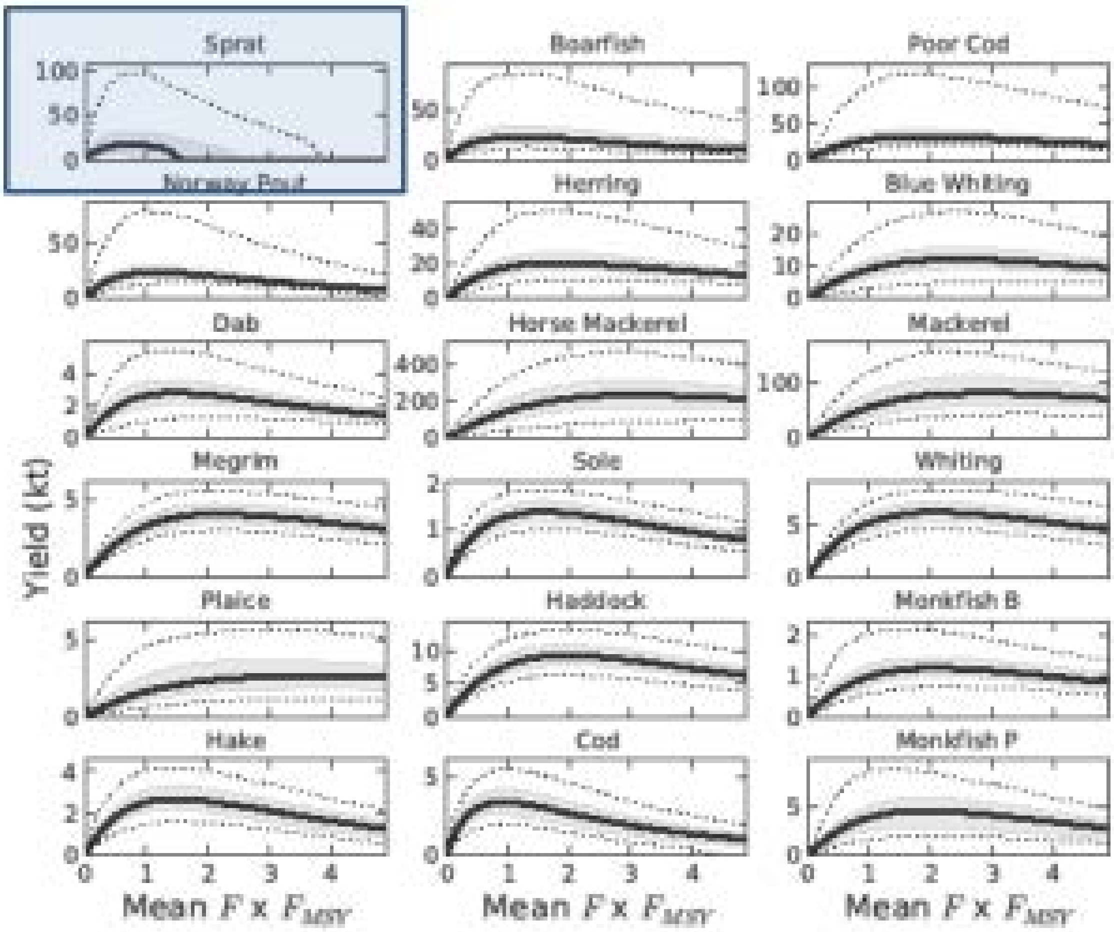

Figure 2.5.1. Yield curves for the 18 Celtic Sea stocks assuming a single unselective fleet.

Four fleets were introduced to the model to allow for an assessment of combined mixed fisheries and multispecies effects. These fleets were otter, beam, and pelagic trawlers, and static gears, and were characterised by reference to the STECF dataset. An initial study of the potential fleet uncertainty is presented in the section on ToR C.

References

Hall, SJ, Collie, JS, Duplisea, DE, Jennings, S, Bravington, M, and Link, J. A length-based multi-species modelfor evaluating community responses to fishing. Canadian Journal of Fisheries and Aquatic Sciences, 63: 1344–1359, 2006

Thorpe, RB, WJF Le Quesne, F Luxford, JS Collie, S Jennings, Evaluation and management implications of uncertainty in a multispecies size-structured model of population and community responses to fishing, Methods in Ecology and Evolution 6 (1), 49-58, 2015

WGSAM. Report of the Working Group on Multispecies Assessment Methods (WGSAM), 9–13 November 2015. Woods Hole, USA:: authors, Daniel Howell, Steve Mackinson, Alexander Kempf, Anna Rindorf, Andrea Belgrano, Robert Thorpe, Morten Vinther, Valerio Bartolino, John Pope, Alfonso Perez Rodriguez, Clement Garcia, Sigrid Lehuta, Isaac Kaplan, Sarah Gaichas, Curti Kiersten, Sean Lucey, Robert Gamble, Harriet Cole, Ulf Lindstrom, Noel Holmgren, Ching Villanueva, Jan Jaap Poos.

2.5.2 Multispecies size spectrum model

As part of the Marine Ecosystem Research Programme, we are developing a Celtic Sea version of the multispecies size spectrum model of Blanchard *et al.* (2014). The model will be fitted with robust measures of uncertainty using methods developed in Spence *et al.* (2016).

Blanchard, J.L., Andersen, K.H., Scott, F., Hintzen, N.T., Piet, G., and Jennings, S. 2014. Evaluating targets and trade-offs among fisheries and conservation objectives using a multispecies size spectrum model. J. Appl. Ecol. 51(3):612–622. doi:10.1111/1365-2664.12238.

Spence, M. A., Blackwell, P. G. and Blanchard, J. L. 2016. Parameter uncertainty of a dynamic multispecies size spectrum model. Can. J. Fish. Aquat. Sci. 73: 589–597. dx.doi.org/10.1139/cjfas-2015-0022

2.5.3 Ecopath in the Irish Sea

No progress updates from 2015 received, although it was noted that there will be Irish Sea benchmark in 2017, which should take ecosystem/multispecies considerations into account.

2.6 Ecoregion F: North Sea

Update on North Sea Model

We have continued to develop a North Sea version of the LeMans model (Hall *et al.*, 2006; Thorpe *et al.* 2015, 2016). The main development relates to methodology for constructing a time-dependent version of the LeMans North Sea model which produces forecasts for specific years. We presented this methodology, with the aim being to test it using a retrospective validation in which the model is tuned to the period 1990–2010 and then evaluated for its ability to “forecast” the period 2010–2015.

The revised methodology is shown in Figure 2.6.1:

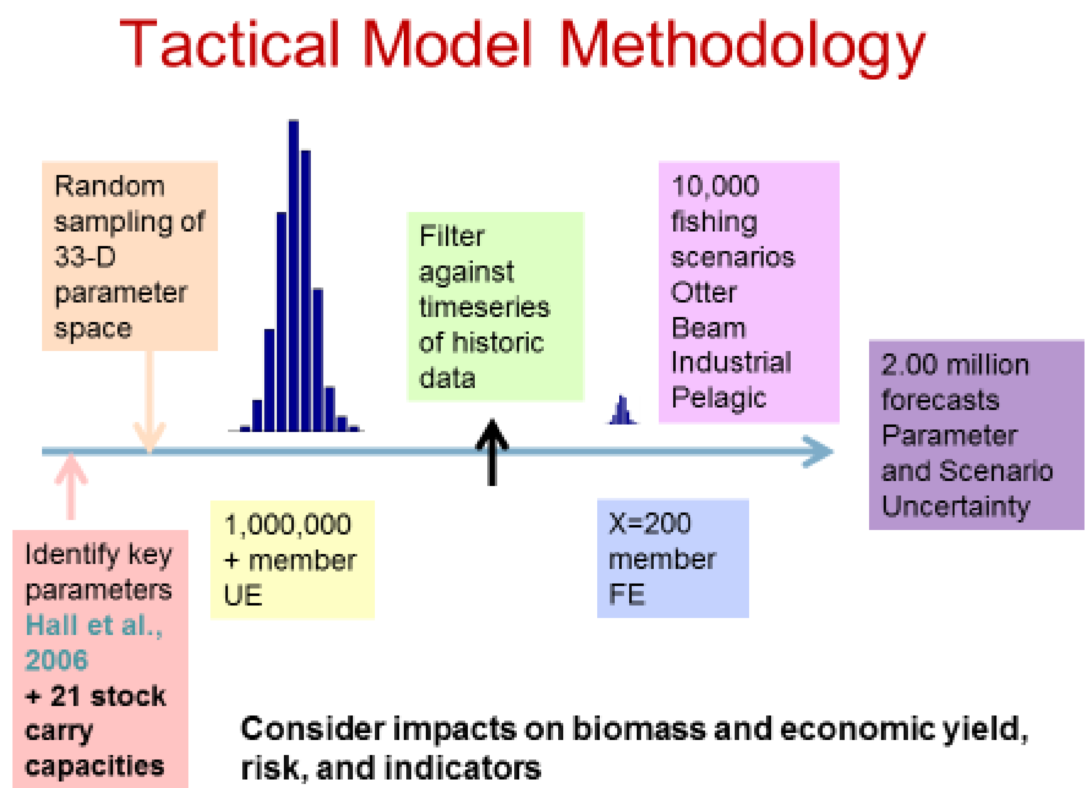

Figure 2.6.1. Schematic of the revised methodology for the time-dependent (transient) model being developed.

The initial parameter space to be explored has increased significantly, driven by the more demanding nature of the screen against data, and the increased probability of rejecting any single parameter combination. We evaluated this space with the aid of a genetic algorithm based upon differential evolution. 2500 initial guesses of the 33-D parameter space in 50 populations of 50 were made, with each parameter value being drawn at random from a uniform distribution. Each candidate solution was scored for its ability to reproduce the ICES assessed biomasses for 10 stocks for each of the 20 years from 1990–2010. Differential evolution was then used to evaluate potential replacement candidates for each of the 2500 guesses. Once this had been done, each population was scored on the basis of its best member, with the worst performing population being replaced by one in which each member was based upon the best member of each of the current populations, subject to a small random perturbation. The process was then repeated 100 times, and a list made of all solutions which scored better than a certain threshold.

Once the genetic screen had completed, the list of "acceptable" outcomes was pruned to eliminate parameter sets that were essentially duplicates of others in the list. Then, starting from the best solution, additional ones were selected from the reduced list, based upon their maximum Euclidian distance from other selected successful parameter choices, until 200 such outcomes had been chosen, and this list of 200 was then used to characterise the successful model ensemble.

Yield curves from this ensemble, for an idealised unselective fleet (c.f. Thorpe *et al.*, 2015) are shown in Figure 2.6.2.

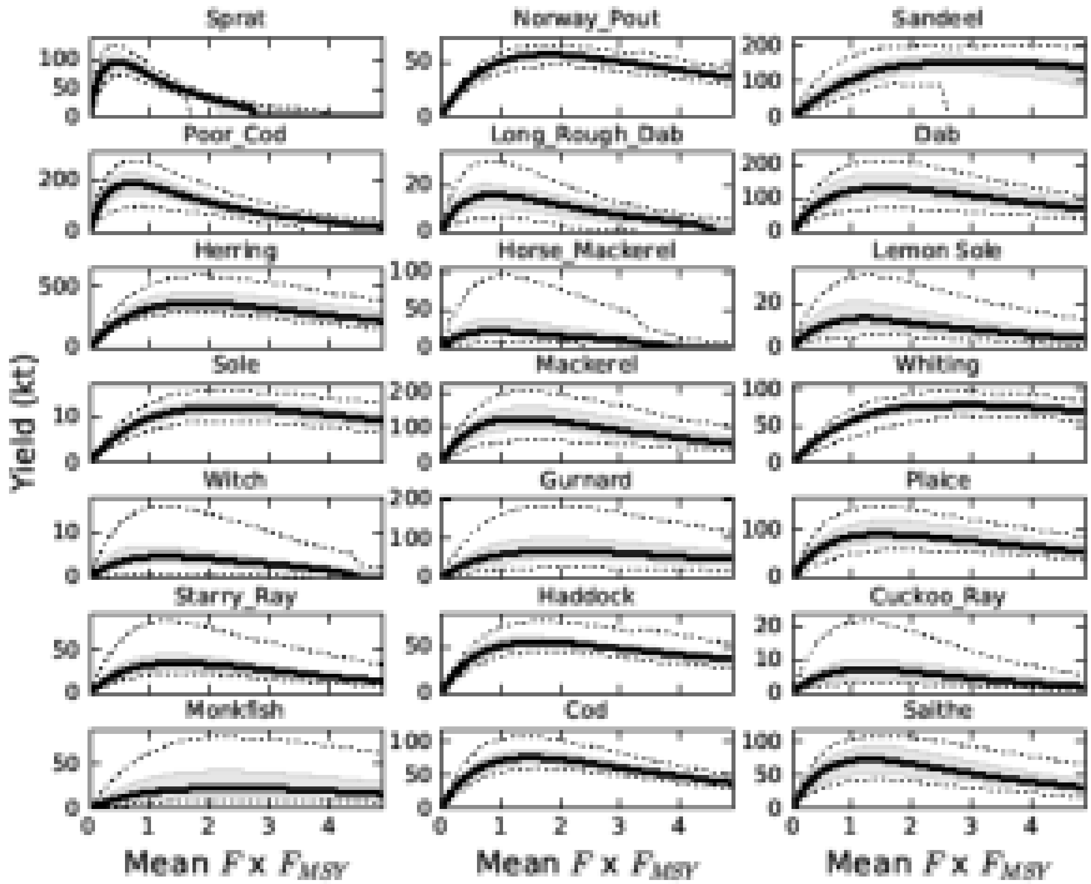

Figure 2.6.2. Yield curves or the 21 North Sea stocks for the time-dependent ensemble of 200 "adequate" models.

Results are qualitatively similar to the model ensemble generated from fits to 1990–2010 average biomasses, the main differences being that the yields are somewhat lower, and the stabilities to fishing by this unselective fleet somewhat higher.

A subsequent discussion of the tuning methodology was very useful and suggested several changes that might improve model performance if they were adopted. The final tuning outcome is shown in Figure 2.6.3:

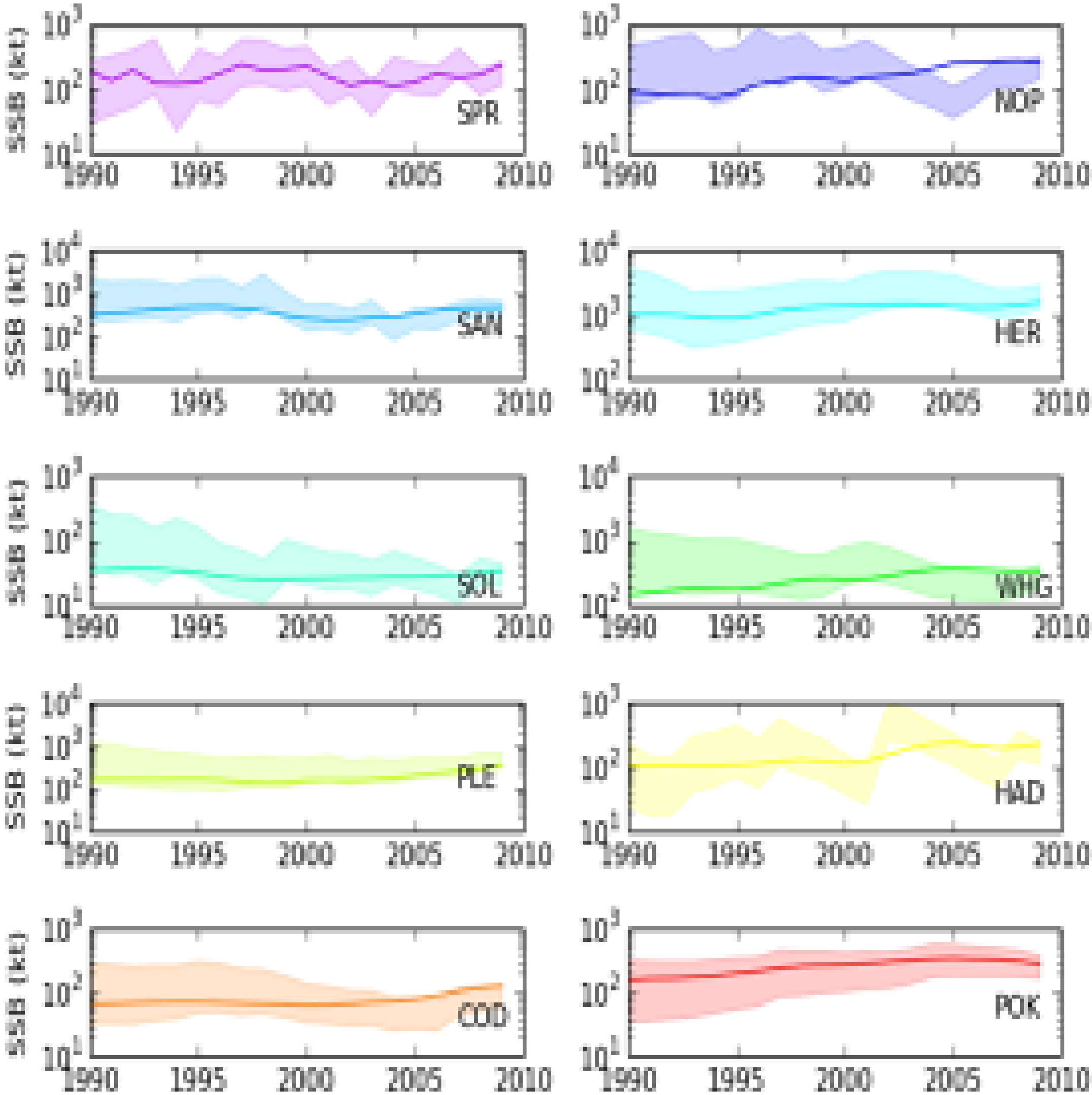

Figure 2.6.3. Spawning stock biomass trajectories for the 10 assessed stocks in the North Sea model. The coloured band are the range of biomasses deemed consistent with the assessments, and the thick lines are the model spawning stock biomass trajectories.

The model is relatively good at getting the biomass scale as it has been tuned for this, but at the expense of variability and sometimes trend (e.g. whiting – green, Norway pout – blue). This result underlines the need to evaluate several aspects of the simulation at the same time, particularly a) scale, b) trend, and c) “bendiness”, whilst ruling out silly solutions in which all of the biomass is in tiny or large size classes.

References

Hall, SJ, Collie, JS, Duplisea, DE, Jennings, S, Bravington, M, and Link, J. A length-based multi-species modelfor evaluating community responses to fishing. Canadian Journal of Fisheries and Aquatic Sciences, 63: 1344–1359, 2006

Thorpe, RB, WJF Le Quesne, F Luxford, JS Collie, S Jennings, Evaluation and management implications of uncertainty in a multispecies size-structured model of population and community responses to fishing, Methods in Ecology and Evolution 6 (1), 49-58, 2015

2.6.1 Moment-based delay difference model in the North Sea

Dr John Pope was unable to attend the meeting but forwarded presentation relating to work done in the MAREFRAME project 'An interactive multispecies model of the North Sea suitable for stakeholders use'. The presentation is available on the WGSAM SharePoint site.

2.6.2 Ecopath with Ecosim for the southern part of the North Sea

A southern North Sea Ecopath with Ecosim (EwE) has been finalized at the Thünen Insitute of Sea Fisheries to a fitted and calibrated stage. An application in identifying multispecies MSY and good environmental status (GES) for the food-web has been published in Ecological Modelling (Stäbler *et al.* 2016), including the model description and its parameterization in the appendix of the manuscript. In the manuscript, we exposed trade-offs between the fleets' objectives and explored, what a possible variant of a multispecies MSY could look like by subjecting the modelled system to a range of different fishing effort levels of the three main fleets (Otter, beam, and brown shrimp trawlers). Long-term projections highlighted multiple fishing regimes that lead to catches of at least 30% of all focal single species MSYs at the same time (see figure 2.6.2.2). Higher simultaneous yields of all four focus species (cod, plaice, sole and brown shrimp) could not be achieved, such that we can assume a risk for the southern North Sea's fisheries that multispecies 'pretty good yields' might fail. Key to the intuitively unsatisfying results are trade-offs between the yields of shrimp fishers and demersal trawlers, where brown shrimp significantly benefit from reduction of its predators cod and whiting, that maximum catches of the shrimp are only achieved when cod are overfished and the yields to the otter trawlers is thus much lower than they could be at 'healthier' cod stocks.

In the next years it is planned to develop an Ecospace model based on the existing Ecosim model. This model will be used to explore spatial management strategies also in relation to the choke species problem due to the landing obligation (i.e. which areas should be closed when to allow a more selective fishing).

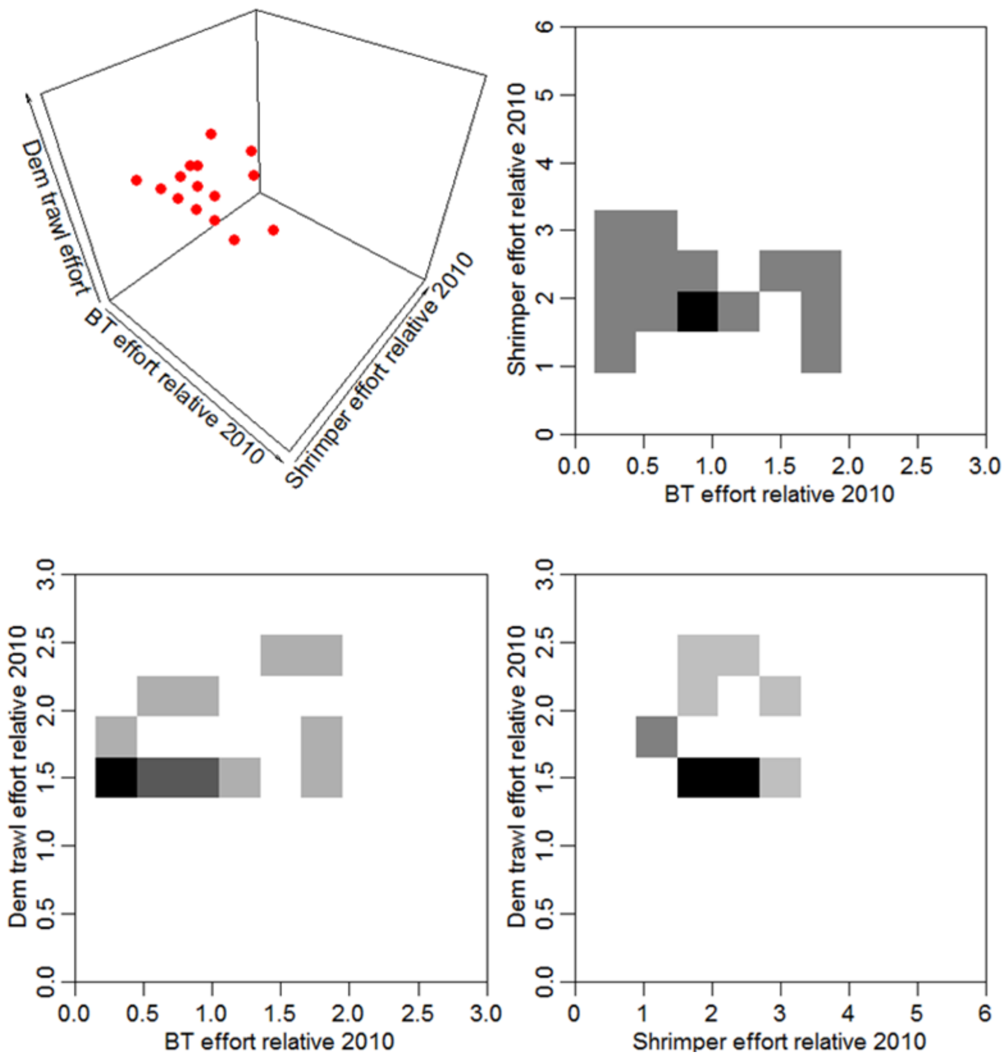

Figure 2.6.2.2. Spheres indicate effort regimes that lead to all four scope species — cod, plaice, sole and brown shrimp – to be simultaneously caught at 30% of their respective maximal possible catches.

2.6.3 Ecopath with Ecosim in the English Channel

A presentation was made on the different EwE models which were developed and/or updated at different zones of the English Channel: Courseulle-sur-Mer, Bay of Seine, Seine estuary, eastern English Channel and Western English Channel. Modelling works being done aim to evaluate zone response to specific perturbations. The model specific to Courseulle-sur-Mer deals with the evaluation of windmill farm development impacts on biological resources and fisheries (Raoux *et al.* 2017; in prep.). The Bay of Seine on benthic habitats and species distribution (Prezy *et al.* In prep.). Modelling Seine estuary sub-zones and network comparison analyses (Tecchio *et al.* 2015). Input data in the English Channel models are being updated based on recent collected data (Villanueva *et al.* In prep) are being undertaken. Spatio-temporal analyses will also be performed at different time and space-scales to test system stability and response to cumulative anthropic perturbations and climate change. Forcing factor data to drive the model will be based on

Chl-a result runs from a hydrodynamic model, MARS3D (Dumas and Langlois 2009), survey catch data and biomasses. Spatial simulations in ECOSPACE will be done in different spatial scales in the Channel.

2.6.4 Medium complexity MSE model for North Sea cod

For North Sea cod Thünen Institute has adapted a single species MSE during a master thesis to be able to take cannibalism and predation from other species on cod into account. The output from the multi species model SMS was analysed and simple relationships between the abundance of the predator and prey could be established. These relationships were implemented in the MSE and e.g., the probability of falling below Blim at different fishing mortalities could be quantified. The MSE was also used to estimate F_{MSY} in a single (constant M) and multi species context. Also the benefit of knowing developments in M with different uncertainty levels associated could be analysed. In the future the MSE will be used to test the current procedure of updating M every 3 years, i.e. under which scenarios this is beneficial or has even negative effects. In the following years it is planned to incorporate simple relationships for various species in FLBEIA, a mixed fisheries model that also includes economics.

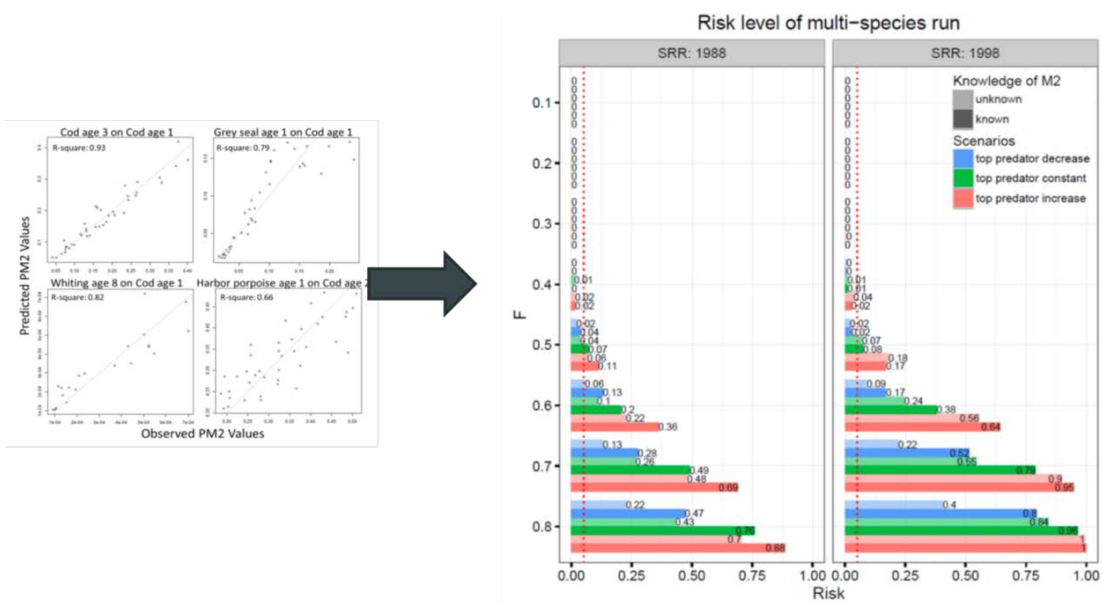

Figure 2.6.4.1. Examples of simple relationships between SMS natural mortality (M) estimates and M values calculated just from predator and prey abundance. These relationships have been implemented in a single species MSE to provide e.g., risk levels under different scenarios.

2.7 Ecoregion G: South European Atlantic Shelf

There is no progress to report on multispecies modelling in the Ecoregion this year.

2.8 Ecoregion H: Western Mediterranean Sea

There is no progress to report on multispecies modelling in the Ecoregion this year.

2.9 Ecoregion I: Adriatic–Ionian Seas

There is no progress to report on multispecies modelling in the Ecoregion this year.

2.10 Ecoregion J: Aegean–Levantine

There is no progress to report on multispecies modelling in the Ecoregion this year.

2.11 Ecoregion K: Oceanic northeast Atlantic

There is no progress to report on multispecies modelling in the Ecoregion this year.

2.12 Ecoregion L: Baltic Sea

An Ecopath with Ecosim key run for the Central Baltic Sea has been presented and is reported under ToR B. An Ecospace extension of that model was briefly presented as well, its validation using survey data and sensitivity analysis is going to be conducted during next year. The model is habitat capacity-driven (Christensen *et al.* 2014). For validation, average environmental conditions from the period 2000–2008 are used as static drivers. The model framework is also set-up for spatio-temporal simulations driven by environmental driver maps changing at an annual time step and Ecosim forcing functions and effort time-series.

Christensen, V., *et al.* (2014) Representing variable habitat quality in a spatial food web model. Ecosystems. 17:8 1397-1412

A Gadget modelling framework is currently under development in the Baltic Sea to analyse the population dynamics of eastern Baltic cod, central Baltic herring and Baltic sprat during the last four decades. A first multispecies model accounting for the effect of predation of cod on the clupeid popu-lations is up running. The model makes use of age and length information from commercial catch and survey data, including both bottom trawl and acoustics. Cod stomach data are also used to characterize cod consumption in the model. Work is in progress on including a feedback of consumption on cod growth. The mechanistic implementation of predator-prey interaction and its consequences on growth offered by Gadget is attractive but poses several challenges such as the entangled effect of different preys (including not explicitly modelled preys, ie benthos), the confounding effect of drivers other than food (ie, temperature), the size-dependency which regulates trophic interactions, other processes such as variable spatial overlap which are not accounted in the current single area model. A multi-area model for cod is also under development with the intention to be extended into a multispecies framework within the next two years. This work is leaded by SLU and carried on within the project MareFrame (EU FP7 #613571).

2.13 Ecoregion M: Black Sea

There is no progress to report on multispecies modelling in the Ecoregion this year.

2.14 Ecoregion: Canadian Northwest Atlantic

Will hold a meeting in November with a goal to "explore methods for operationally incorporating the ecosystem approach into single-species stock assessments and advice.

Provide at least one concrete example of how it could be implemented in Canada (proof of concept)".

2.15 Ecoregion: US Northwest Atlantic

Sean Lucey is continuing to develop Rpath, an R package implementing static and dynamic food web models using the same basic equations as Ecopath with Ecosim. The purpose of developing this software is to provide a reproducible, multi-platform food web model to enhance simulation, visualization, and customization of analyses and outputs available from the basic EwE models. The manuscript describing the model structure and function is undergoing final edits and should be submitted within the next month. The software is publically available at https://github.com/slucey/RpathDev . To install using devtools::install_github include ref = 'Public', e.g.:

devtools::install_github('slucey/RpathDev/Rpath', ref = 'Public', build_vignettes = TRUE)

Sean welcomes questions and feedback on the software. Two workshops have been held to date (Seattle, December 2015 and Woods Hole, June 2016) to further develop the R package and assist users with specific applications. On the Northeast US shelf, existing food web models implemented for four subregions by Link *et al.* in the early 2000s ("EMAX" models) will be updated and implemented in Rpath with more recent data and disaggregated species groups. These models will be used as MSE operating models.

The Northeast US Atlantis model (NEUS Atlantis) is undergoing an update to disaggregate key commercial species and modernize the parameterization to take full advantage of the current Atlantis codebase. This update should improve the model's utility as an MSE operating model for the region. Version 1.5 of Atlantis has now been parameterized and will undergo calibration within the coming months.

In August, 2016, the NOAA North Atlantic Regional Team (NART) sponsored a working group advance the systematic treatment of animal movement in NOAA ecosystem and assessment models. A working group of agency and academic partners held 10 pre-workshop conference calls with several presentations over the course of six months (March to August) culminating in a 3-day workshop on 15–17 August 2016 at the University of Massachusetts, Dartmouth Center for Innovation and Entrepreneurship. The purpose of the conference calls and workshop was to consider and review:

1) Types of animal movement into, out of, and within the NES LME and the management implications of including or not including these movements in analyses and models;
2) Incorporation of movement parameters/transition matrices into ecosystem and stock assessment models;
3) Data available and needed to adequately address movement for given management questions;
4) Environmental and habitat drivers that influence movement.

From this review, the AMWG developed a hierarchy of best practices for incorporating movement into models based on the management needs, data availability, and models available, and began to apply these best practices for including movement into selected models as case studies. Planned products from the AMWG include a NOAA Technical Report, multiple manuscripts, and updated ecosystem and assessment models.

The National Marine Fisheries Service established a national Management Strategy Evaluation (MSE) working group in late 2015. The group is intended to foster communication among regional Science Centers and build MSE capacity nationwide. Each Center will have or has hired a dedicated MSE position, with expertise ranging from fish and protected species stock assessment to ecosystem modelling to economics. A list of current MSE projects is being compiled across Centers. Sarah Gaichas is a co-chair of this group and welcomes contacts from all working on MSEs.

There is a project ("Poseidon") in progress in the US to generate test datasets for multi-species and single species model performance testing. The project will use Atlantis, an end-to-end ecosystem model, to build ecological datasets representing the "truth" in a skill assessment of six increasingly complex fisheries assessment models from single species through full food webs (single-species biomass dynamics, single-species age structured, multi-species biomass dynamics, multi-species age structured, full food web biomass dynamics, and full food web age structured). Both estimation and forecasting/MSE skill of these models will be assessed and compared across a range of environmental and human use scenarios. Two teams are envisioned to participate: Team Poseidon will use two existing Atlantis models to create simulated ecosystems with multiple climate and anthropogenic forcing scenarios, and will define a set of skill assessment performance criteria. Team Odysseus will consist of several sets of experienced fisheries modelers, each assigned to one or more of the tested models. Team Odysseus will first use the baseline "data" generated by Team Poseidon to build each of the 6 test models for each of the two simulated ecosystems (a total of 12 models), and calibrate and/or fit the model to the baseline data according to standard practice for each model. Results will then be compared to the "truth" represented by the Atlantis runs to assess model skill under different conditions.

At present the Poseidon project has collected and developed tools to extract information representing the "truth" from Atlantis models, and has developed prototype tools to create simulated "survey" and "fishery" datasets with known properties for input into assessment models. Many international collaborators contributed to these tools in a workshop helpd prior to the Atlantis Summit of December 2015 (Weijerman *et al.*, 2016). Tools in development are available at https://github.com/r4atlantis/atlantisom . Ultimately, users will specify survey areas, seasons, target species, catchabilities, size selectivities, and the desired level of observation error to develop survey datasets. Fishery datasets will be built in a similar manner. Simulating diet composition data for input into EwE and similar models is also in progress. The project is being done without direct funding, but these tools will be completed as time allows, and the tentative plan is to have a workshop with Team Odysseus within the next year.

Additional progress in the Ecoregion is reported under ToRs E, F, and G.

Weijerman, M., Link, J. S., Fulton, E. A., Olsen, E., Townsend, H., Gaichas, S., Hansen, C., *et al.* 2016. Atlantis Ecosystem Model Summit: Report from a workshop. Ecological Modelling, 335: 35–38.

2.16 Ecoregion: Southern Shelf Seas

There is no progress to report on multispecies modelling in the Ecoregion this year.

References

Dumas F, Langlois G. 2009. Model for Applications at Regional Scale: Scientific model description. IFREMER, Plouzané. 103 pp.

Raoux A., *et al.* 2017. Benthic and fish aggregation inside an offshore wind farm: Which effects on the trophic web functioning? Ecological indicators 72: 33-46.

Stäbler, M., *et al.* 2016. "Combining efforts to make maximum sustainable yields and good environmental status match in a food-web model of the southern North Sea." Ecological Modelling. 331. 17-30.

Stäbler, M., *et al.* 2014. Potatoes of opportunity for fishing in the southern North Sea: Using Ecopath with Ecosim to identify MSY and GES options. Fisheries Centre Research Reports 2014 Volume 22 Number 3. 22: 29.

STECF 2015. Scientific, Technical and Economic Committee for Fisheries (STECF) – Evaluation of management plans: Evaluation of the multi-annual plan for the North Sea demersal stocks (STECF-15-04). 2015. Publications Office of the European Union, Luxembourg, EUR XXXX EN, JRC XXXX, 152 pp. http://stecf.jrc.ec.europa.eu/web/stecf/ewg1502

Tecchio, S., *et al.* 2015. The mosaic of habitats of the Seine estuary: Insights from food-web modelling and network analysis. Ecological Modelling 312: 91-101.

3 ToR B: Update of key runs

Review of LeMans model

A review of the LeMans model was conducted at WGSAM 2016. There was no formal written request, nor ToRs for this review, and WGSAM therefore decided that the appropriate aim of the was to advise ICES WKIRISH 3 (scheduled for January 2017) regarding the appropriate uses of the LeMans multispecies model ensemble in its current state, and to suggest potential revisions to the framework prior to its presentation at WKIRISH 3 and 4. The specific Irish Sea LeMans model should then be reviewed formally as part of the WKIrish4 process. In addition to the LeMans model, there may also be an EwE model up and running for WKIRISH 4, so multimodel inference could be used to 1) provide ecosystem context for the advice and 2) potentially inform mortality, recruitment, and other single-species stock assessment parameters that are uncertain for some Irish Sea stocks. The LeMans length-based model was reviewed as a multispecies ensemble modelling framework, rather than a particular parameterization for the Irish Sea (which is not yet available). We stress that WGSAM reviewed the LeMans framework as parameterized for the North Sea, since the Irish Sea isn't done yet, and we were therefore not able to give specific comments on the Irish Sea parameterization of the model. WGSAM also didn't have any documentation, model or examples in advance, so this review is based on materials presented at the meeting and ensuing questions and discussion. WGSAM appreciates the efforts of Robert Thorpe in presenting the framework and in soliciting feedback on all aspects of the model.

The LeMans model ensemble is adapted from Hall *et al.* (2006) and Rochet *et al.* (2011). Changes made include replacement of Ricker recruitment with a hockey stick recruitment function to stabilize the model (simpler and easier to understand than B-H), more inter-stock interactions, improved discretisation of selectivity and community metrics, and revectorized to improve speed (x20). The model lies on a spectrum somewhere between a MICE-SMS style tactical model fitted to data for all stocks, and a strategic Ecopath style model. It includes specific size structured stocks, intermediate complexity, multispecies model, and is top down (predation) focused – and therefore not well suited to bottom-up investigations (although work is ongoing to investigate adding prey-dependent growth to the model). Parameters governing predator size selection, natural mortality (M1), spawner-recruit relationships, growth efficiency, carrying capacity, and asymptotic length were varied to construct the unfiltered ensemble reflecting reasonable uncertainty ranges for these parameters. Then, ensemble members were tested against biomass time-series from ICES assessments for major stocks and filtered down to a set of parameter sets producing "reasonably" fitting models in order to form the filtered ensemble for further analysis as in (Thorpe *et al.*, 2015).

First, WGSAM emphasizes that the LeMans model ensemble is not intended to provide tactical annual quotas for modelled species. We also stress that as an ensemble modelling scheme, LeMans is best suited to providing "plausible ranges" rather than point estimates. Further, prior to the January 2017 meeting it will not be sufficiently coordinated with single species models to provide natural mortality (M2) time-series for use in single species assessments (as SMS does in the North Sea).

Based on its review, WGSAM recommends use of the LeMans model ensemble as a contextual model, which is best used to provide advice on multispecies reference points and on the potential range of, and trends in, predation mortality in the Irish Sea ecosystem under different fishing scenarios. For example, the LeMans ensemble seems particularly suited to evaluating the potential performance of particular fleet combinations as specified by Irish Sea experts and stakeholders, after the optimization is done to narrow the potential range of parameterizations in the ensemble. One could ask, for example, what is the Nash equilibrium giving multispecies MSY, and then ask how achievable that would be with existing or modified fleets targeting subsets of species with specific size selectivities?

To ensure that the LeMans framework can meet this objective, WGSAM recommended the following improvements:

1) Extend criteria for retaining parameter sets to encompass matching biomass scale, trend, and variability. This should ensure that the selected ensemble will perform better in matching the biomass trends of the species. In addition, add criteria evaluating population demographic parameters such as average length or weight to ensure that unrealistic size distributions are not retained within the feasible parameter sets.

2) Ensure that fixed parameters for LeMans (length-weight parameters, growth parameters, etc.) are compatible with/identical to those used in single species assessments to be reviewed under WKIRISH.

3) Evaluate the evidence for the scale and possible trends in additional mortality on fished species from unmodelled marine mammals, seabirds, and other

sources. If these are large and variable then they may need to be included as drivers in the model.

4) Estimate "other food" as a parameter rather than fixing it outside the model, and evaluate whether system production may have changed during the modelled period.

5) Evaluate whether there is evidence within the Irish Sea sufficient to change the default assumptions in the diet matrix and potentially improve from the current presence/absence (1/0) format. For example, do species overlap for only part of a year, suggesting that the interaction matrix value could be multiplied by an overlap factor?

6) Evaluate whether changes in environmental conditions, for example temperature, in the Irish Sea may violate the assumptions of stationary physical processes in the model, and if such drivers could be included in some of the physical processes modelled (e.g. growth and consumption).

7) Critically important to test if the length discretization is appropriate to the stocks included in the model (i.e. if the results are sensitive to the choice of discretization).

Given that the criteria for acceptable fits ("what does success look like?") determine the subset of parameterizations retained within the LeMans ensemble, WGSAM focused on suggestions to improve fit criteria beyond the currently presented biomass ranges, where parameter sets were considered equally valid if modelled biomass time-series were within a factor of +-3 relative to ICES stock assessment biomass early in the hindcast model run to +- 1.5 late in the hindcast model run. While this resulted in biomass levels reasonably near those estimated by current ICES assessments, trends were not matched for all species, and other characteristics may be of interest as well. Fits to summary statistics were suggested as a method for including multiple characteristics in the fit criteria as in Approximate Bayesian Computation (ABC). Overall, statistics addressing the scale of biomass as well as trends and variance were recommended. For example, Pearson correlations or the simple slope could be calculated to assess fits to trend. The variance of the difference between the fit line and the model output could be used to assess pattern. Finally, a statistic such as fit to mean length or weight in the population could be used to assess whether the size structure produced by a given parameter set in the modelled population was realistic – i.e. providing a filter to avoid biologically implausible solutions. In all cases, avoiding overfitting the ICES stock assessments was considered important—there are legitimate reasons for a multispecies model having different results from single species models, which are not themselves observations of a system but rather alternative interpretations.

After these recommendations are considered, WGSAM looks forward to a further review of the model as parameterized and optimized for the Irish Sea at the 2017 WGSAM meeting. Following that review, WGSAM may recommend that M2 distributions estimated by the LeMans ensemble could be used qualitatively in comparison with single species assessment models. For example, do the single species natural mortalities fall within the feasible range or outside? Are trends in natural mortality apparent in LeMans ensembles that should be considered in single species assessments? Similarly, recruitment series from the LeMans ensemble may be useful in identifying where there are key predation impacts for consideration in single species models. However, WGSAM does not recommend direct transfer of a particular realization of the LeMans ensemble natural mortality or recruitment time-series for a particular species to the single species assessment: this would not be an appropriate use of this type of ensemble model. Rather, the LeMans ensemble can provide valuable information on potential trajectories of the entire system considering multiple types of uncertainty that are not considered in single species assessments.

References

Hall, S. J., Collie, J. S., Duplisea, D. E., Jennings, S., Bravington, M., and Link, J. 2006. A length-based multispecies model for evaluating community responses to fishing. Canadian Journal of Fisheries and Aquatic Sciences, 63: 1344–1359.

Rochet, M.-J., Collie, J. S., Jennings, S., and Hall, S. J. 2011. Does selective fishing conserve community biodiversity? Predictions from a length-based multispecies model. Canadian Journal of Fisheries and Aquatic Sciences, 68: 469–486.

Thorpe, R. B., Le Quesne, W. J. F., Luxford, F., Collie, J. S., and Jennings, S. 2015. Evaluation and management implications of uncertainty in a multispecies size-structured model of population and community responses to fishing. Methods in Ecology and Evolution, 6: 49–58.

Baltic Sea- Ecopath with Ecosim

An Ecopath with Ecosim model of the open Baltic Sea based on an earlier model (Tomczak *et al.*, 2012, Niiranen *et al.* 2013) has been modified and fit to time-series data from 2004-2013 (Bauer *et al., in prep*). It was presented as a first “key-run” for the Baltic. The ‘draft’ key run was presented in plenary with subsequent evaluation of the inputs, outputs and documentation being reviewed by a subgroup of experts in discussion with the modeller. Aspects of the Ecopath model reviewed in greater detail included Ecopath biomass, production/biomass and consumption/biomass parameters, the resulting mortality rates, the fleet definitions and PreBal diagnostics. For the Ecosim component, the group mostly discussed selection and weighting of time-series data used in fitting and the setting of vulnerability parameters. The group also reviewed parameters like feeding time adjustment rates, noting that the default value of 0.5 may not be appropriate for all groups. Specific recommendations for the model are documented below.

Baltic Sea Model, 2004-2013

Key run summary sheet

Area	Baltic Sea
Modelling approach	Ecopath with Ecosim
Type of model	Foodweb compartment
Run year	2016
Species/Groups	22 functional groups
Time range	2004-2013
Time-step	Yearly (internal multistanza calculations monthly)
Area structure	Model covers approximately Baltic Proper ICES Subdivisions 25-29, excl. 28-1. A spatial extension of the model is under development.
Stomach data	From the EU tender “Study on stomach content of fish to support the assessment of good

	environmental status of marine foodwebs and the prediction of MSY after stock restoration"
Purpose of key run	Description of changes in the Baltic Sea foodweb
Model changes since last key run	First key run

The fit of model predictions to biomass data from surveys or assessments for all higher trophic level species is shown in Figure 3.1. Changes in selected system and community indicators are shown in Figure 3.2.

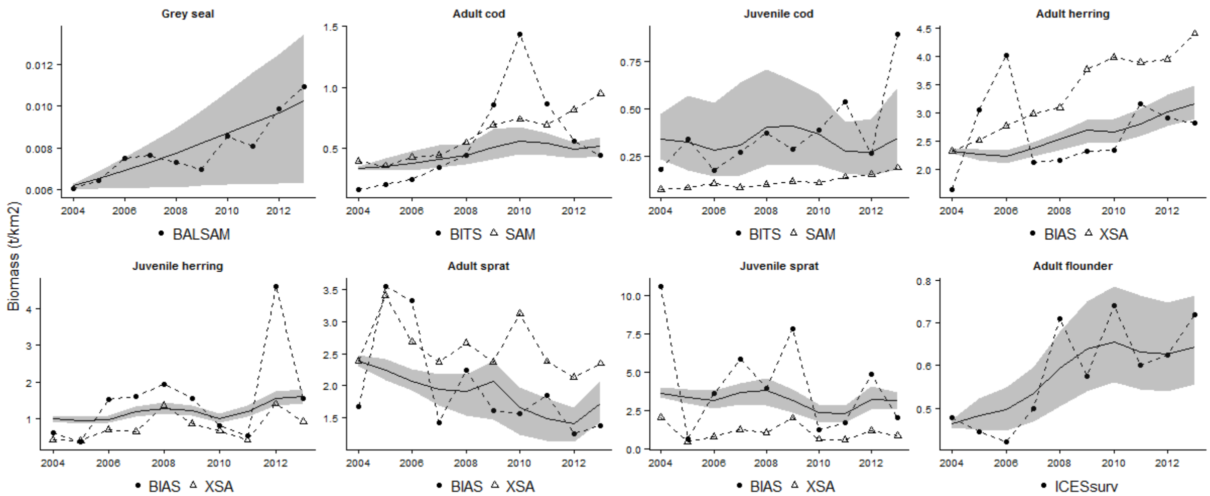

Figure 3.1. Ecosim predicted biomasses of higher trophic level species (solid lines) fit to time-series of surveys (black circles, dashed lines) and time-series of assessments (empty triangles, dashed lines) after final calibration of vulnerability multipliers. Fish survey indices of abundance (BITS, BIAS, ICESsurv) are rescaled for visualization. Grey areas indicate 90% confidence intervals based on Monte Carlo simulations varying Ecopath input biomasses, P/B and Q/B parameters, biomass accumulation parameter of grey seals and ecotrophic efficiency of mysids.

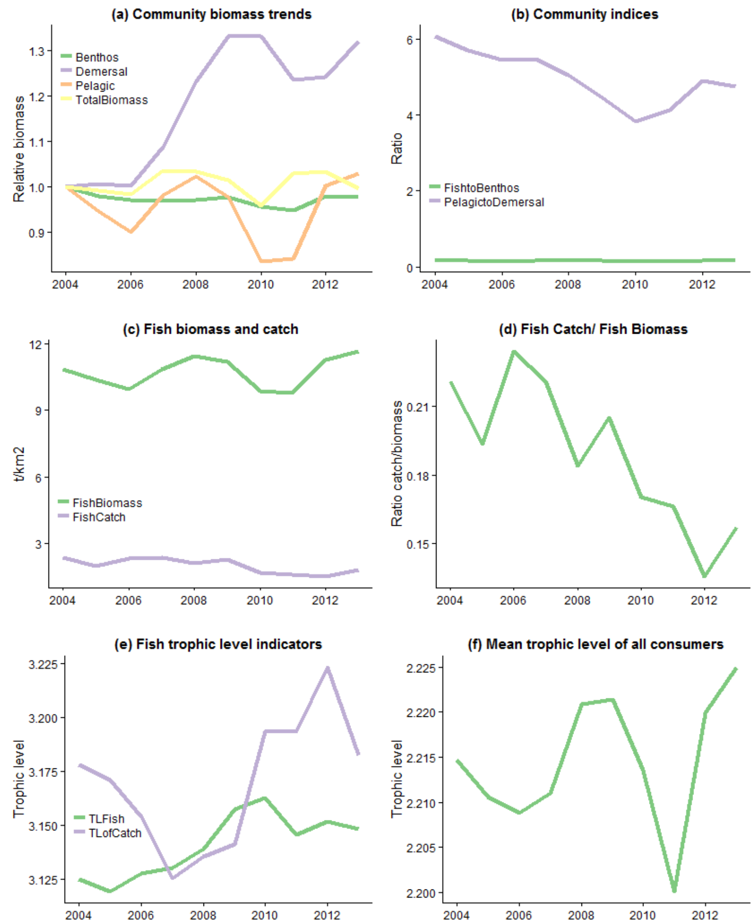

Figure 3.2. Ecosim indicators derived from the model key run. For more details see Annex 3.

The most important issues raised during group discussions were:

- Similar to the North Sea key run last year (WGSAM 2015), the P/B ratio of cod (*Gadus morhua*) applied in the model was considered to be too high compared to expected values based on the species' size and lifespan. Similar to last year, this was based on a high fishing mortality value derived from data. The mass-balance requirement in Ecopath implies that a species' production is high enough in the Ecopath starting year to be in balance with fishing and natural mortalities: P/B=total mortality (Z)=fishing mortality (F)+natural mortality (M). Similarly, the P/B ratio for herring, which is harvested at a relatively low rate in 2004, may be too low. In further work it needs to be investigated how a high P/B ratio affects stock recovery scenarios projected by models. More details about this issue are described in the 2015 WGSAM report (p. 48) for the North Sea key run.
- Sprat biomass was indicated to be too high in the key run draft. PreBal algorithms based on Link (2010) show that sprat biomass was relatively high compared to its zooplankton prey. In addition, there was a fairly large unaccounted production within the model (small EE). This is related to how biomass estimates by stock assessments were converted to biomass densities.

Fish biomass calculations were revised and are reported in Annex 3, Table 1. Sprat biomass still remains relatively high compared to zooplankton. However, the post-regime shift food web of the central Baltic that the model represents is indeed known to be dominated by sprat, which suppresses zooplankton biomass (Möllmann *et al.*, 2008; Casini *et al.*, 2009). Besides, sprat biomass density has become highly spatially heterogeneous during the last decades (Casini *et al.*, 2014), which complicates the interpretation of simple energetic balance tests such as the ones applied in PreBal.

- Discard estimates for juvenile Baltic cod were found to be too low. We revised discard estimates of cod by using ICES data instead of data reported by the European Commission's Scientific, Technical and Economic Committee (STECF) as was done previously.
- Juvenile cod are fished at a low level and predated upon little in the model, thus, their 'other mortality' term is high. This was considered to be questionable and the group suggested to review relevant information available. This has been done with the following results. The low level of fishing mortality in the draft key run was a result of using unreliable STECF age-based data for estimating discards as noted before. After revision of discard estimates, fishing mortality increased by a factor of 10, but still remained very small, probably reflecting the size-selectivity of Baltic fisheries. Predation mortality of juvenile cod is also small, as its two predators, seals and large cod were at relatively low density compared to smaller, juvenile cod around the model year 2004 (ICES, 2015). Thus, the major mortality source is 'other mortality', which is consistent with studies showing juvenile cod survival to be related to environmentally driven food availability (Huwer *et al.*, 2014).
- There was a long group discussion on the use of information from surveys, assessment biomass estimates, and catches and how each should be weighted relative to one another. The group did not come to a consensus and the overall suggestion was to try different weightings and combinations of time-series, and investigate the sensitivity of the resulting vulnerability estimates, which was done (see Annex 3 for details).
- It was suggested during discussions that the parameter 'Fraction of other mortality sensitive to changes in feeding time' should be set to 0 for seals, birds and cod instead of the default 1 to keep their 'other mortality' insensitive to their feeding time. We followed the suggestion for seals, as their major sources of mortality (disease, pollution) are independent on their feeding time. However, we set the parameter 0.5 for birds and adult cod. One of the major sources of mortality for fish-feeding birds is getting trapped in fishing gear when foraging for food, thus, this mortality has a relationship with feeding time. In addition, setting this parameter to 0 for a certain group in general increases its recovery time after a decrease in biomass (Blanchard *et al.*, 2002). During model testing we noted that in the case of birds this meant a recovery time of several hundred years after an approx. 30% reduction in biomass, which we found too extreme. Adult cod 'other mortality' rate is also likely to be linked to foraging time. Most important sources of 'other mortality' for this group are probably parasitism (Horbowy *et al.*, 2016) and low condition (Dutil

& Lambert, 2000; Horbowy, 2016) and the latter is likely further deteriorating if cod has to forage longer for prey or dive to anoxic waters more often when prey availability is low (Hinrichsen *et al.*, 2011).

References

Bauer B, Bartolino V, Stenbeek J, Saraiva S, Orio A, Tomczak MT Spatial effects of eutrophication on Baltic Sea fish based on their environmental tolerance and foodweb interactions. *in prep*.

Blanchard JL, Pinnegar JK, Mackinson S (2002) Exploring marine mammal-fishery interactions using "Ecopath with Ecosim": modelling the Barents Sea ecosystem. *Sci. Ser. Tech. Rep.*, 52.

Casini M, Hjelm J, Molinero J-C *et al.* (2009) Trophic cascades promote threshold-like shifts in pelagic marine ecosystems. *Proceedings of the National Academy of Sciences of the United States of America*, **106**, 197–202.

Casini M, Rouyer T, Bartolino V, Larson N, Grygiel W (2014) Density-dependence in space and time: Opposite synchronous variations in population distribution and body condition in the baltic sea sprat (Sprattus sprattus) over three decades. *PLoS ONE*, **9**, 1–9.

Dutil JD, Lambert Y (2000) Natural mortality from poor condition in Atlantic cod (Gadus morhua). *Canadian Journal of Fisheries and Aquatic Sciences*, **57**, 826–836.

Hinrichsen HH, Huwer B, Makarchouk A, Petereit C, Schaber M, Voss R (2011) Climate-driven long-term trends in Baltic Sea oxygen concentrations and the potential consequences for eastern Baltic cod (Gadus morhua). *ICES Journal of Marine Science*, **68**, 2019–2028.

Horbowy J (2016) Effects of varying natural mortality and selectivity on the assessment of eastern Baltic cod (Gadus morhua Linnaeus, 1758) stock. *Journal of Applied Ichthyology*

Horbowy J, Podolska M, Nadolna-Ałtyn K (2016) Increasing occurrence of anisakid nematodes in the liver of cod (Gadus morhua) from the Baltic Sea: Does infection affect the condition and mortality of fish? *Fisheries Research*, **179**, 98–103.

Huwer B, Hinrichsen HH, Böttcher U, Voss R, Köster F (2014) Characteristics of juvenile survivors reveal spatio-temporal differences in early life stage survival of Baltic cod. *Marine Ecology Progress Series*, **511**, 165–180.

ICES (2015) Report of the Benchmark Workshop on Baltic Cod Stocks (WKBALTCOD), 2-6 March 2015, Rostock, Germany. 172.

Möllmann C, Müller-Karulis B, Kornilovs G, St John M a. (2008) Effects of climate and overfishing on zooplankton dynamics and ecosystem structure: Regime shifts, trophic cascade, and feedback loops in a simple ecosystem. *ICES Journal of Marine Science*, **65**, 302–310.

4 ToR C: Consider methods to assess the skill of multispecies models intended for operational advice

4.1 Some leads toward improved confidence and transparency in WGSAM models

One reason why multi species and ecosystem models are so far hardly used in fisheries management is a general perception that the models are highly uncertain and their skill to predict stock dynamics is limited. Many people still feel that single species approaches are more reliable although ignoring the dynamic of important processes like predation.

Therefore, a skill assessment will be undertaken by WGSAM in the next two years to challenge available multi species models from different regions and compare their performance to single species approaches. An outline of work that needs to be conducted has been discussed during WGSAM 2016.

As a starting point each member of the group will collect examples from his/her region where multi species approaches have been used already and have worked in fisheries management. Also examples where it would have been better to use a multi species approach will be collected. The aim is to get an overview on which is the best tool for which task (e.g., short term predictions to set TACs vs. long-term predictions to estimate reference points, evaluation of trade-offs).

Next to this overview the possibilities to test the skill of models on three different levels have been discussed:

1) One of the main outputs from multi species models are estimates of natural mortality M. In order to be able to estimate the right M the diet selection models are most important to predict diet compositions for years where no stomach data are available. Therefore, predicted diet compositions will be confronted with observed stomach data from different regions. For example, Kempf *et al.* 2010 could show that the diet composition of North Sea cod and whiting can be explained to a large extent by the relative abundance of prey items and the spatial overlap between predator and prey. Such an approach could be extended to other regions (i.e. regions where a time-series of stomach data is available) but also simple comparisons between observed and predicted diet compositions are possible. The aim is to demonstrate that it is possible to get reliable results with the current process understanding even if annual stomach data are not available. However, also situations where the current process understanding is not sufficient will be highlighted.

2) In many regions MS models have been fitted to historical data (hindcasts) over quite a long period. It is possible to shorten the hindcasts and predict the stock dynamics for the remaining years with and without time dynamic M. The trend in predictions can be compared to fishery independent surveys to test whether multi species models can outperform single species approaches in forecasting stock dynamics and in which situations. Because F has decreased for predator species in several regions in the last 10 years, this should have impacted the food webs. Predictions will range from 2 to 10 years to test the performance on different time scales. The predictions will be also forced with catch and recruitment information separately to see which processes have the highest impact on the performance of the predictions and determine stock dynamics.

3) MSE-performance testing with operating models of different complexity (from medium complexity to end-to end models) is possible in various regions. With an MSE approach the performance of single species and multi species assessments can be tested with different assumptions on uncertainties and bias. Also the current procedure of using M estimates from multi species models in single species assessments can be evaluated. Here also the influence of time lacks (e.g., keyrun North Sea every 3 years) can be tested. If available, in such an ap-

proach also the performance of modelling ensembles could be tested and whether they outperform single models (also link to ToR D).

Intersessional work for the WGSAM meeting in 2017 is planned especially on levels 1 and 2 in various regions (e.g., North Sea, US East Coast, Barent Sea). The meeting in 2017 will be used to present first results and agree on further steps needed to finalize the work in 2018.

References

Kempf A, Floeter J, Temming A (2008) Predator-prey overlap induced Holling type III functional response in the North Sea fish assemblage. Mar Ecol Prog Ser 367:295-308, DOI:10.3354/meps07555

5 ToR D: Investigate the performance of multi-model ensemble in comparison to single model approach

5.1 A dynamic multi-model ensemble for ecosystem simulators

The ensemble model, developed as part of the Marine Ecosystem Research Programme, was presented which aims to combine outputs from different marine ecosystem models, both multispecies and single species, in order to make inferences about the real world. The model, based on the ideas developed by Chandler (2013), treats the outputs from different marine ecosystem models as coming from a population that centers on the simulator consensus, which is itself not the truth but a bias version of it which is learnt.

One of the major difficulties in applying these ideas is that marine ecosystem models have different outputs and are on different scales, for example in Strathclyde End to End (Heath, 2012) species are aggregated by their living habitat whereas in LeMans (Thorpe, 2015) the species are modelled explicitly. The ensemble model uses correlations in other ecosystem models to determine what the models that group species would have predicted for individual species, for example what Strathclyde End to End would predict for sole given its prediction for demersal species. A proof of concept case study was demonstrated where the ensemble model examined what would happen to demersal fish if we were to stop fishing in the North Sea.

Chandler, R. E. 2013. Exploiting strength, discounting weakness: combining information from multiple climate simulators. Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences, 371(1991)

Heath, M. R. 2012. Ecosystem limits to food web fluxes and fisheries yields in the North Sea simulated with an end-to-end food web model. Progress in Oceanography, 102:42-66, 2012.

Thorpe, R.B., Le Quesne, W.J.F., Luxford, F., Collie, J.S., and Jennings, S. 2015. Evaluation and management implications of uncertainty in a multi-species size-structured model of population and community responses to fishing. Methods Ecol. Evol. 6(1): 49–58.

6 ToR E: Test performance and sensitivity of ecosystem indicators

6.1 Food-web evenness, a novel biodiversity indicator

Biodiversity has aesthetic, functional and intrinsic values. Maintaining or restoring biodiversity is an essential goal of numerous management policies and it is target of international agreements such as the Convention on Biological Diversity (CBD, 2010).

Simulation of alternative management scenarios using ecosystem models and estimation of potential consequent impacts on ecosystems is becoming common practice and new ecosystem models and scenarios are developed every year for a larger number of areas and ecosystems. However, we identified strong practical and theoretical limitations in the use of many currently available biodiversity indicators on ecosystem model output.

Species richness-based indices have been proved to be highly useful for analysing scenarios in global or large-scale species distribution studies, but they are expected to perform poorly when simulating management scenarios within single ecosystems, over geographically limited areas and within the temporal scale of one or two decades where the number of species is unlikely to change and most typical responses are changes in species relative biomasses. So called 'umbrella species' are supposed to be indicative of diversity within food-webs but their choice is necessarily subjective and they discard a large part of the information which is provided by ecosystem models. Indices of evenness may be expected to be more suitable than other indicators when it comes to their application on ecosystem model output, but they make sense and can be applied as a measure of diversity only within one trophic level or functional group. When applied to a whole food-web, the expectation of evenness among taxa does not hold. Large part of energy is lost when it is transferred between trophic levels and as a consequence, higher trophic level species are expected to be less abundant than lower trophic level ones according to the 'pyramid of biomass' concept.

We developed an evenness index which accounts for the loss of energy and biomass towards higher trophic levels. The index takes high values if 1) biomasses compared among trophic levels decrease according to a pyramid and 2) species and functional groups at the same trophic-level have even biomasses. These two criteria are used to generate an expected distribution of biomasses for the modelled system. Then similarity to this theoretical expectation is used to measure the ecosystem state in relation to when neither a specific species within a trophic level, nor a specific trophic level is disproportionally abundant.

We applied the food-web evenness index on EwE model output from two different ecosystems, i.e. the Baltic Sea (Tomczak *et al.*, 2012) and the West coast of Scotland (Alexander *et al.*, 2015), and on surveys data conducted in the North Sea in 2000/2001 using stable isotopes information to derive trophic levels (Jennings and Blanchard, 2004). Application of the index on the Baltic Sea model output were presented and discussed at WGSAM.

This indicator is expected to be suitable for analysing simulated management scenarios and inform about the ecosystem state as ecosystems are more likely to be unstable and their function disrupted when biomass at a certain trophic level strongly declines (Carpenter *et al.*, 1985; Prugh *et al.*, 2009) and when one or few species dominate a trophic level (Atkinson *et al.*, 2014).

Alexander, K.A., Heymans, J.J., Magill, S., Tomczak, M.T., Holmes, S.J., Wilding, T.A., 2015. Investigating the recent decline in gadoid stocks in the west of Scotland shelf ecosystem using a foodweb model. ICES J. Mar. Sci. 72, 436–449.

Atkinson, A., Hill, S.L., Barange, M., Pakhomov, E.A., Raubenheimer, D., Schmidt, K., Simpson, S.J., Reiss, C., 2014. Sardine cycles, krill declines, and locust plagues: Revisiting "wasp-waist" food webs. Trends Ecol. Evol. 29, 309–316. doi:10.1016/j.tree.2014.03.011

Carpenter, S.R., Kitchell, J.F., Hodgson, J.R., 1985. Cascading trophic interactions and lake productivity. Bioscience 35, 634–639.

CBD, 2010. Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity at its Tenth Meeting, Decision X/2, in: The Strategic Plan for Biodiversity 2011-2020 and the Aichi Biodiversity Targets. Secretariat for the Convention on Biological Diversity, Nagoya, Japan.

Jennings, S., Blanchard, J.L., 2004. Fish abundance with no fishing: Predictions based on macroecological theory. J. Anim. Ecol. 73, 632–642. doi:10.1111/j.0021-8790.2004.00839.x

Prugh, L.R., Stoner, C.J., Epps, C.W., Bean, W.T., Ripple, W.J., Laliberte, A.S., Brashares, J.S., 2009. The rise of the mesopredator. Bioscience 59, 779–791. doi:10.1525/bio.2009.59.9.9

Tomczak, M.T., Niiranen, S., Hjerne, O., Blenckner, T., 2012. Ecosystem flow dynamics in the Baltic Proper-Using a multi-trophic dataset as a basis for food-web modelling. Ecol. Modell. 230, 123–147. doi:10.1016/j.ecolmodel.2011.12.014.

6.2 Identifying ecosystem thresholds & evaluating indicator performance (US)

Sarah Gaichas presented work in progress by Jamie Tam, and reviewed previous work by Scott Large and Gavin Fay using both empirical and modelling approaches to estimate ecosystem threshold responses to different drivers. A challenge in working with ecosystem indicators is that both ecosystem drivers and ecosystem responses can be complex and multidimensional. A single ecosystem indicator may be difficult to relate to an individual driver, because processes operating at different scales and multiple drivers may complicate responses. Here, several multivariate methods were applied to empirical datasets representing multiple ecosystem drivers and responses across four US marine ecosystems: the Eastern Bering Sea, the California Current, the Gulf of Mexico, and the Northeast US. Gradient forests (Ellis *et al.*, 2012) were used to evaluate whether common drivers acting together might lead to common multivariate responses in ecosystems, and whether ecosystem thresholds could be identified using multiple indicators responding to multiple drivers. Then, dynamic factor analysis was used to further characterize and analyze ecosystem trends, and generalized additive models were used to illustrate another method to identify threshold responses to individual drivers such as fisheries landings across the four ecosystems. Thresholds identified with these multiple methods were reasonably robust within an ecosystem. While outcomes of these empirical methods are somewhat dependent on time-series length and quality, the overall methods are promising for integrating multiple drivers and evaluating cumulative effects of both human activities and environmental pressures. Several of these approaches were first applied to empirical data for the Northeast US shelf ecosystem (Large *et al.*, 2013, 2015). Further, threshold responses have been simulated in modelled fish populations (Fay *et al.*, 2013) with additional work on the performance of indicator-based control rules (Fay *et al.*, 2015).

Ellis, N., Smith, S. J., and Pitcher, C. R. 2012. Gradient forests: calculating importance gradients on physical predictors. Ecology, 93: 156–168.

Fay, G., Large, S. I., Link, J. S., and Gamble, R. J. 2013. Testing systemic fishing responses with ecosystem indicators. Ecological Modelling, 265: 45–55.

Fay, G., Link, J. S., Large, S. I., and Gamble, R. J. 2015. Management performance of ecological indicators in the Georges Bank finfish fishery. ICES Journal of Marine Science: Journal du Conseil, 72: 1285–1296.

Large, S. I., Fay, G., Friedland, K. D., and Link, J. S. 2013. Defining trends and thresholds in responses of ecological indicators to fishing and environmental pressures. ICES Journal of Marine Science: Journal du Conseil, 70: 755–767.

Large, S. I., Fay, G., Friedland, K. D., and Link, J. S. 2015. Quantifying Patterns of Change in Marine Ecosystem Response to Multiple Pressures: e0119922. PLoS One, 10. http://search.proquest.com/docview/1664222461/abstract/120C1207C54548BEPQ/1 (Accessed 14 October 2016)

6.3 Indices of community status using a Species-Area-Relationship (SAR) model

Novaglio, C., Svedäng, H., Sköld, M., Belgrano. A.

Indices of community status, SAR (Species-Area Relationships), LFI (Large Fish Index) and MTL (Mean Trophic Level) where tested for the Swedish west coast including Skagerrak and Kattegat for assessing marine ecosystem status and outcomes of management actions such as Marine Protected Areas (MPAs) and trawl limit regulations. The SAR slope (i.e. rate at which species accumulates with increasing area) and the MTL showed to be significant ecosystem status indices for mapping the fish community recovery and provide information on functional biodiversity. The LFI showed high variability and no significant increase after management enforcement, suggesting that this index is too linked to the abundance of dominating species. The SAR approach provides novel insights in the way we perceive and understand changes at the community and ecosystem level in relation to biodiversity loss, fishery and governance (Novaglio *et al.* 2016 *submitted)*. The SAR slope may be viewed as a novel indicator to be further considered and explore within the EU MSFD descriptors D1 and D4.

Novaglio, C., Svedäng, H., Sköld, M., Belgrano**.** A. Species-Area Relationship (SAR) and Marine Protected Areas (MPAs): linking fish communities status and marine conservation measures. *NATURE Scientific Reports submitted* (2016).

7 ToR F: Meta-analysis of impact of top predators on fish stocks in ICES waters

Northeast US mammal/fishery simulations

Sarah Gaichas presented simulation work by Laurel Smith and others (Smith *et al.*, 2015b) evaluating potential tradeoffs between marine mammals, fish, and fisheries on the Northeast US shelf. Laurel had previously estimated the total consumption of fish by marine mammals on the shelf to be greater than or equal to fisheries catch for many taxa

(Smith *et al.*, 2015a). Consumption of a single species, herring, by all fish predators combined was approximately twice fisheries catch (Overholtz and Link, 2007). Therefore, investigating potential conflicts between fisheries management, predator consumption, and marine mammal recovery is of interest here. A multispecies production model (Gamble and Link, 2009) simulated marine mammal/fish/fishery interactions over a range of fishing rates and with alternative assumptions regarding the dependence of marine mammals on fish prey and the level of human-caused mortality on mammals. The model was parameterized to allow for predation, competition, and prey feedback interactions, as well as fisheries catch and incidental mortality of marine mammals resulting from bycatch and ship strikes. Marine mammals and fish were aggregated into general taxonomic guilds to evaluate general interactions. Guild biomass status relative to either a Bmsy proxy (fish) or current biomass (mammals) varied by fishing scenario, with the highest levels of fishing causing poor status for the targeted fish. However, marine mammal recovery rates were only affected under both high fishing scenarios and the assumption of increased incidental mortality relative to observed. Within the current fishing scenario, the relative effects of predation, fishing, or prey loss varied by guild, with all fish dominated by predation mortality rather than fishing mortality, but with baleen and toothed whales dominated by human caused mortality rather than prey loss. Finally, trajectories and fishery reference points compared under current fishing were insensitive to assumptions about marine mammal mortality or prey dependence for pelagics, but more sensitive for groundfish and flatfish. Overall, the simple model was considered useful for evaluating potential tradeoffs between marine mammal and fisheries management objectives, and showed that marine mammals were both affected by potential changes in vessel interactions and to a lesser extent prey availability through fisheries management, and in turn affected fishery reference points.

Gamble, R. J., and Link, J. S. 2009. Analyzing the tradeoffs among ecological and fishing effects on an example fish community: A multispecies (fisheries) production model. Ecological Modelling, 220: 2570–2582.

Overholtz, W. J., and Link, J. S. 2007. Consumption impacts by marine mammals, fish, and seabirds on the Gulf of Maine–Georges Bank Atlantic herring (Clupea harengus) complex during the years 1977–2002. ICES Journal of Marine Science: Journal du Conseil, 64: 83–96.

Smith, L. A., Link, J. S., Cadrin, S. X., and Palka, D. L. 2015a. Consumption by marine mammals on the Northeast U.S. continental shelf. Ecological Applications, 25: 373–389.

Smith, L., Gamble, R., Gaichas, S., and Link, J. 2015b. Simulations to evaluate management tradeoffs among marine mammal consumption needs, commercial fishing fleets and finfish biomass. Marine Ecology Progress Series, 523: 215–232.

Discussion outlining work for upcoming years

Diet information from elasmobranchs available in DAPSTOM

An investigation of the DAPSTOM database held by CEFAS revealed that there is diet information for various elasmobranchs (e.g. see Figure 7.1). However, at least in the public downloads information is only given about how often each prey was found in the stomachs and in how many hauls. Information by weight is missing what gives a serious bias towards small prey. In addition, the number of stomachs is often low and distributed over various areas and years. Therefore, these data have to be use carefully and only a rough overview on the diet is possible. CEFAS will be asked whether additional information is available that cannot be downloaded. The information available may be used to get an idea whether in the ecosystems mainly piscivorous sharks have been replaced by sharks eating more benthos. Hypothesis can be formulated what this change has caused in the ecosystems and it can be tested in ecosystem models what a recovery may mean for the food webs and yield from commercially important fish.

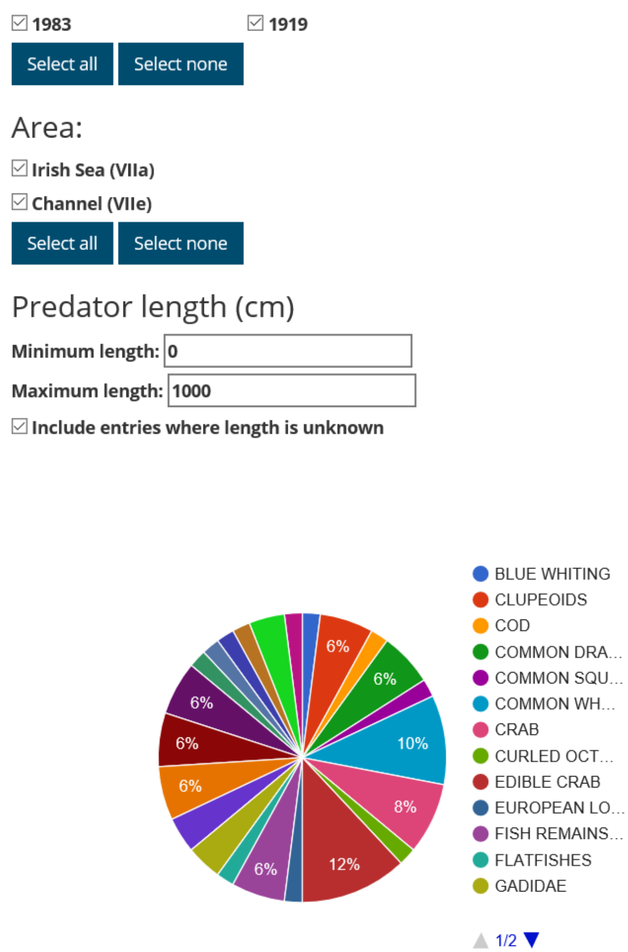

Figure 7.1. Example of diet information available for greater spotted dogfish.

8 ToR G: Explore the consequences of multispecies, mixed fisheries interactions and environmental factors in practical multispecies management (MSY related and other biological reference points)

8.1 ECOSYSTEM-Fmsy project

This is a forthcoming proposed project, which is currently partly funded, aiming at collating multispecies Fmsy values for stocks in a range of ecoregions, using production models as a starting point.

WGSAM can support the project as valuable, and considers that it would be able to build upon existing region reviews and model results to give a wider synthesis, and potentially represent a step towards including multispecies considerations into practical management. The group would **not** recommend relying on single-species production models for this work, but would consider a wider overview of Fmsy estimates as valuable. Standard single species production models have fishing morality as only process that changes stock dynamics over time (i.e. lower catch leads to an increase in biomass). They are neither able to handle trends in recruitment nor natural mortality and show a poor performance if other processes than fishing mortality dominate stock dynamics. Using models that can handle other processes than fishing mortality seem to be a prerequisite for the project to achieve an overview over the range of uncertainties. While single species production models may be the best available in data poor regions, this is not the case for the regions in this project which are both data and model rich, and the best available range of estimates would be of interest.

8.1.1 The common fisheries policy and Nash equilibrium MS-MSY

On the WGSAM meeting 2015, a Nash equilibrium (NE) among stock harvest rates were proposed as multi-species (MS) MSY reference point(s) (ICES, 2016b). The NE-MSYs were then calculated for the pelagic stocks of the eastern Baltic Sea using MSI-SOM model (Norrström *et al.*, 2016). This consists of one prey species, cod, and two clupeid species. The NE-FMSYs were found too high compared to SS-MSYs, and particularly for the predator the mortality rate was nearly doubled. The high MS-FMSYs was similar to the values obtained from the SMS model, which is another MS-model of the Baltic Sea. Our experience in a meeting with one of the EU commissioners was that the MS-FMSYs were hard to accept for them, even though it is common for MS-models to produce higher FMSYs of the higher trophic layers compared to SS-models.

The NE-FMSYs were produced with MSY as a target in accordance with ICES, however, the common fisheries policy of the EU (European Commission, 2013) article 2(2) states:

"Therefore, the Union should improve the CFP by adapting exploitation rates so as to ensure that, within a reasonable time-frame, the exploitation of marine biological resources restores and maintains populations of harvested stocks above levels that can produce the maximum sustainable yield."

The interpretation of this paragraph is that MSY is not a target, it (BMSY) is a limit and the stocks should be managed to stay above BMSY (Veitch *et al.*, 2015):

"Fishing consistently at FMSY will not fulfill the MSY objective in Article 2(2), because on average it will only result in stocks being above levels that can produce MSY half of the time. Therefore, and in line with the United Nations Fish Stocks Agreement, which the EU is a signatory to, FMSY must be implemented as a limit, not a target, exploitation rate."

ICES regards MSY as a target and does not use BMSY as a reference point (ICES, 2015):

"The ICES approach to advice on fishing opportunities integrates the ecosystem and precautionary approach with the objective of achieving maximum sustainable yield (MSY).

The (MSY-) approach does not use a BMSY estimate. BMSY is a notional value around which stock size fluctuates when fishing at FMSY."

Furthermore, ICES suggest that F should not be reduced until the stock biomass falls below levels that are to be expected from fishing at FMSY:

"MSY Btrigger is considered the lower bound of spawning–stock biomass fluctuation around BMSY. It is a biomass reference point that triggers a cautious response. The cautious response, in cases where the spawning stock falls below MSY Btrigger, is to reduce fishing mortality to allow a stock to rebuild to levels allowing for MSY."

If the variation in the SSB around the BMSY is substantial, the difference in perspectives on the CFP will result in quite different levels of F-targets, SSBs and perhaps in yields. We ran the MSI-SOM for the two scenarios, (i) the FMSY as a target, and (ii) the SSB lower 5% percentile = BMSY (we will use the acronym BAB for this target). We ran a genetic algorithm to find the Fs associated with (ii). We set a harder punishment for the 5% percentile being below rather than above the BMSY. The reason being that it may not be possible to keep SSB percentiles of all stocks at BMSY, and then we prefer one or two stocks being a little bit above rather than below.

We encountered some variation between runs trying to find the BAB so the precise numbers are preliminary (Table 8.1.1). The cod F is reduced considerably and very close to the value of 0.3 that was in the previous management plan (European Commission, 2007). For herring, the BAB-F is 0.25, which is closer to the current SS-FMSY of 0.22 of the stock (ICES, 2016a). The BAB-F for sprat of 0.25 is also much closer to the SS-FMSY of 0.26 for sprat. The average SSBs is higher for all three species. Herring was below Bpa at the NE-MSY, but the BAB elevated the average SSB above Bpa. The yields increase somewhat for the cod, decreases for the herring and increases for sprat. Sprat exhibits tripled yield and average biomasses.

Table 8.1.1. A comparison of F, SSB and yield (Y) between the two management objectives: MSY as a target and SSB>BMSY (BAB). The MSY refers to the Nash equilibrium of multispecies MSY. Blim and Bpa are from ICES advice 2016 (no limits are available for cod).

Nash Equilibrium for multispecies MSY (NE-MSY)								
	F		Avg. SSB				Yield	
	MSY	BAB	MSY	BAB	Blim	Bpa	MSY	BAB
Cod	0.47	0.31	211	268	n.a.	n.a.	76	78
Herring	0.3	0.25	460	615	430	600	115	90
Sprat	0.54	0.33	794	2520	410	570	402	1247

The BAB may produce more appealing results within the MS-MSY framework, in this case the NE-MSY. If the BAB-approach was to be applied to single species quotas these would likely decrease compared to the current SS-MSYs as well. Some species may reach abundances that may not be desirable, in this case one of the forage fish species. These levels need to be checked for wider ecosystem consequences, for instance in an Ecopath with Ecosim model.

European Commission 2007. COUNCIL REGULATION (EC) No 1098/2007 of 18 September 2007 establishing a multiannual plan for the cod stocks in the Baltic Sea and the fisheries exploiting those stocks, amending Regulation (EEC) No 2847/93 and repealing Regulation (EC) No 779/97.

European Commission 2013. Regulation 1380/2013/EU on the Common Fisheries Policy, amending Council Regulations (EC) No 1954/2003 and (EC) No 1224/2009 and repealing Council Regulations (EC) No 2371/2002 and (EC) No 639/2004 and Council Decision 2004/585/EC.

ICES 2015. ICES Advice basis. *In* Advice Book 1. ICES, Copenhagen.

ICES 2016a. The Baltic Sea. *In* Advice Book VIII. ICES, Copenhagen.

ICES. 2016b. Report of the Working Group on Multispecies Assessment Methods (WGSAM), 9–13 November 2015, Woods Hole, USA. ICES CM 2015/SSGEPI:20. 206 pp.

Norrström, N., Casini, M., and Holmgren, N. M. A. 2016. Nash equilibrium can resolve conflicting maximum sustainable yields in multi-species fisheries management. ICES Journal of Marine Science, DOI: 10.1093/icesjms/fsw148.

Veitch, L., Luk, S., and Tacconi, F. 2015. MSY in the CFP: A Legal Briefing, ClientEarth, London.

8.1.2 The approach for ecosystem-based management in the Northeast United States

Under the authority of the Magnuson-Stevens Fisheries Management and Conservation Act, the United States has established eight regional management councils. The Northeast Fisheries Science Center provides scientific support for two of those councils, the New England and Mid-Atlantic Fishery Management Councils (NEFMC and MAFMC respectively). Each council is taking a slightly different approach to including ecosystem and multispecies effects in their management.

New England

The NEFMC has set up an ecosystem-based fisheries management plan development team (EBFM PDT) tasked with developing a draft fisheries ecosystem plan (FEP). The EBFM PDT has developed a strategy of setting an overall system level cap on removals with individual species protection. The International Commission on Northwest Atlantic Fisheries (ICNAF) had implemented a similar management system in 1973 in which an upper cap on removals from U.S. waters ranging from Cape Hatteras to the Gulf of Maine was established such that total species-level quotas could not exceed the system cap. Additional constraints related to by-catch levels in each fishery were also imposed (for a review see Hennemuth and Rockwell 1987).

The first step in implementing the strategy is to define the spatial footprint of the ecosystem. The NEFSC has identified four ecological production units based on biogeophysical characteristics of the region (Fogarty *et al.* 2011). For their draft prototype FEP, the NEFMC selected the Georges Bank EPU. Georges Bank is a highly productive submarine plateau located due east of Massachusetts. We then apply a hierarchical approach in which system level production is determined and allocations among fishery functional groups (defined below) specified.

The overall cap can be set using a simple energy flow model to determine ecosystem production potential (Fogarty *et al.* 2016) or other modelling approaches. To define a limit ecosystem reference point, it has been proposed that removals from the system do not exceed the proportion of microplankton total production as this is the major pathway for energy to the higher trophic levels. Once the overall system cap has been determined, it will be allocated among functional groups such that the sum of the catches from the functional groups does not exceed the system-level ceiling on removals.

We define a Fishery Functional Group (FFG) as species that are caught together by specified fleet sectors and that play similar roles in the ecosystem with respect to energy transfer. Because the species are caught together, they typically share similar habitat use patterns and often size structures related to gear selectivity characteristics. The concept accordingly encapsulates information on technological interactions as well as trophic guild structure and feeding interactions.

The final step involves specifying catch levels for Individual species comprising the functional groups. For this purpose, a quadratic programming algorithm is employed in which the sum of the catches of species within the functional group is maximized subject to the constraint that the functional group cap is not exceeded and individual species biomass does not fall below specified threshold. Accordingly, individual species will be monitored to ensure that none are exploited at unsustainable levels.

Mid–Atlantic

Sarah Gaichas reviewed efforts to integrate species, fleet, climate, and habitat interactions into fishery management with the US Mid-Atlantic Fishery Management Council (MAFMC). The MAFMC approach retains the current single species Fishery Management Plans (FMPs), but incorporates relevant climate, habitat, predator, and other interactions as possible. The NOAA Northeast Fisheries Science Center supported development of the Mid-Atlantic Fishery Management Council (MAFMC) Ecosystem Approach to Fisheries Management (EAFM) policy guidance document over the past several years by working closely with MAFMC Council members and staff. In August 2016 MAFMC formally adopted the document (http://www.mafmc.org/eafm/). This policy guidance document will allow the MAFMC to enhance current management with ecosystem and social science advances and to consider coordinated management of previously separately managed resources. The MAFMC has already initiated action to protect unmanaged forage fish as a result of the EAFM effort (http://www.mafmc.org/actions/unmanaged-forage). The EAFM policy guidance document provides a framework for the Council to develop and consider other regulatory actions addressing specific ecosystem issues.

The framework for integrating ecosystem approaches into current fishery management was built on aspects of the Integrated Ecosystem Assessment (IEA) approach (Levin *et al.*, 2009, 2014). Strategic tools including risk assessment, conceptual modelling, and management strategy evaluation (MSE) are components of the framework (Gaichas *et al.*, 2016), providing the Council with a process for considering these more complex interactions that go beyond the traditional single-species management. Work is in progress on a risk assessment evaluating all of MAFMC's managed species within all Fishery Management Plans (FMPs) against fishery, ecosystem, climate, regulatory, economic, and social risks. Based on the results of this assessment, the Council can prioritize FMPs, species, and or issues to further develop regulatory actions. Conceptual models for key relationships between climate, habitat, ecosystem, social, and economic factors surrounding priority species or FMPs will then be developed to ensure that all important interactions are considered in any analysis. Analyses will be conducted using an MSE framework where the Council and its stakeholders identify objectives and performance measures to evaluate proposed management strategies, and scientists from relevant disciplines collaborate to develop operating models addressing key uncertainties. Ultimately, the framework aims to provide decision support for the Council by identifying tradeoffs between objectives and evaluating the potential performance of any proposed management strategy that considers key interactions, risks and uncertainties.

Fogarty, M.J., R.Gamble, K. Hyde S. Lucey, C. Keith, 2011. Spatial Considerations for Ecosystem-Based Fishery Management on the Northeast U.S. Continental Shelf. In (D. Packer, Ed.) Proceedings of the Mid-Atlantic Management Council's Habitat-Ecosystem Workshop, NOAA Tech. Mem. NMFS-F/SPO-115 pp 31-33.

Fogarty, M.J., Rosenberg, A.A., Cooper, A.B., Dickey-Collas, M., Fulton, E.A., Gutiérrez, N.L., Hyde, K.J.W., Kleisner, K.M., Kristiansen, T., Longo, C., Minte- Vera, C.V., Minto, C., Mosqueira, I., Osio, G.C., Ovando, D., Selig, E.R., Thorson, J.T., Ye, Y., 2016. Fishery production potential of large marine ecosystems: A prototype analysis. Environmental Development 17, Supplement 1, 211-219.

Gaichas, S., Seagraves, R., Coakley, J., DePiper, G., Guida, V., Hare, J., Rago, P., *et al.* 2016. A Framework for Incorporating Species, Fleet, Habitat, and Climate Interactions into Fishery Management. Frontiers in Marine Science, 3. http://www.frontiersin.org/marine_ecosystem_ecology/10.3389/fmars.2016.00105/abstract

Hennemuth, R.C. and S. Rockwell. 1987. History of fisheries conservation and Management. In (R. Backus, Ed.). Georges Bank. MIT Press. Cambridge MA. pp. 430-446.

Levin, P. S., Fogarty, M. J., Murawski, S. A., and Fluharty, D. 2009. Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. PLoS Biology, 7: 23–28.

Levin, P. S., Kelble, C. R., Shuford, R. L., Ainsworth, C., deReynier, Y., Dunsmore, R., Fogarty, M. J., *et al.* 2014. Guidance for implementation of integrated ecosystem assessments: a US perspective. ICES Journal of Marine Science: Journal du Conseil, 71: 1198–1204.

8.1.3 New England Herring MSE

Sarah Gaichas reviewed a management strategy evaluation (MSE) in progress to test harvest control rules that consider Atlantic herring's role as forage in the Northeast US shelf ecosystem. This work was initiated at the request of the New England Fisheries Management Council (NEFMC) in January 2016. This may be the first MSE in the US Fishery Management Council process to hold a public stakeholder workshop to generate objectives and performance measures. An outside facilitator was selected to facilitate a 2-day stakeholder workshop to develop a list of MSE management objectives and performance measures and to identify key sources of uncertainty to be considered in the analysis. The stakeholder workshop was held 16–17 May in Portland, ME, with about 70 attendees representing the fishing industry (commercial herring, lobster, tuna, and groundfish, as well as recreational and for hire), environmental NGOs, Council members and staff, and state, federal, and academic scientists. Workshop materials (http://www.nefmc.org/calendar/may-16-17-2016-herring-workshop) and a workshop report are available (http://s3.amazonaws.com/nefmc.org/6a_MSE-workshop-draft-summary-report.pdf).

Development of simulation tools is proceeding this summer and fall. In general, the short timeframe does not permit development and use of a full ecosystem model (the existing Northeast US Atlantis and EwE models both lacked specific elements required to evaluate key stakeholder objectives). Therefore, a series of simpler models are being linked together. A herring stock dynamics operating model has been parameterized to represent 8 potential states of nature (including combinations of uncertainties in natural mortality, recruitment steepness, herring weight at age, and assessment bias). Six control rule types are being tested: biomass based with four configurations for implementing a particular catch (every year, every 3 years, every 5 years, etc.) and catch based with two configurations. There are 1360 combinations of control rule attributes for biomass based types and 10 attributes for the catch based control rules. Only one server has been filled up so far.

Based on stakeholder input at the May workshop, we plan on linking 4 relatively simple delay-difference predator models to the herring model output, as well as a simple economics model relating herring price to yield. Preliminary analysis with existing EwE models in the region was conducted to frame the predator model parameterization. Predator types include a tuna, a nesting seabird, a groundfish, and a marine mammal. The tuna model is configured based on western Atlantic bluefin tuna population dynamics. Herring affect tuna growth in this model as that was the only impact with available evidence (Golet *et al.*, 2015); no relationship between herring population (biomass or recruitment) and tuna biomass or recruitment was detected. A nesting seabird model is currently in development based on the measured reproductive success, population levels, and chick provisioning diet recorded at managed colonies within the Gulf of Maine. Herring will be linked to reproductive success for seabirds. Groundfish and marine mammal models have yet to be developed, but will likely be based on Atlantic cod and minke whale population dynamics and any available data within the region.

MSE products are expected to contribute to the Council decision making process for Herring FMP Amendment 8 by early 2017.The New England Council is currently planning to schedule a second stakeholder workshop to review results of the herring MSE in December 2016.

Golet, W., Record, N., Lehuta, S., Lutcavage, M., Galuardi, B., Cooper, A., and Pershing, A. 2015. The paradox of the pelagics: why bluefin tuna can go hungry in a sea of plenty. Marine Ecology Progress Series, 527: 181–192.

8.1.4 Nash equilibrium to understand productivity in a multispecies system during environmental change – The Baltic Sea as a case study

Nash equilibrium FMSYs are determined by the ecological interactions between the species in a multispecies system and the environmental drivers (Norrström *et al.* 2016). This makes it possible to evaluate the productivity of the multispecies system at the Nash equilibrium for different environmental scenarios. We investigated the effects of the climate change scenarios of salinity and temperature in the Baltic proper until 2099 from the report by Meier *et al.* (2012). A scenario of reproductive volume (RV) with a reduced RV was investigated. Data of RV from 1981 to 2009 was fitted to a gamma distribution where a reduction in the RV was accomplished by decreasing the mean of the gamma distribution from 150 to 75 and 0, whereas the variance remained constant. The different RV conditions had very little effect so the results are omitted here.

The Nash equilibrium is based on the system reaching an equilibrium so instead of simulating the system with trends in the environmental data we evaluated the system at different points along the way towards, and including, the full 2099 scenario. For salinity and temperature the steps towards the full environmental change in 2099 was divided into 3 steps (linear change) so scenarios were run in current condition, 1/3, 2/3 of the way, and at the full scenario. The productivity of adults and recruits was analyzed where adult productivity represents the growth in biomass of the adult population and recruit productivity is the biomass gained from recruitment. Recruitment age was 2 for cod and 1 for the clupeids.

The environmental changes were shown to have little effect on the productivity of the sprat stock (<10% increase) but stronger effect on the cod (~50% reduction) and especially the herring stock (~80% reduction); (Figure 8.1.4). However, the ratio of adult to recruit productivity seems to be constant throughout the scenarios. These simulations illustrate how the Nash equilibrium can be used to simulate environmental scenarios with an objective fisheries management strategy.

This means that cod productivity decreases with 1% per year and herring productivity with 2% per year. Considerations should be taken whether this should affect the annual recommendations of fishing opportunities.

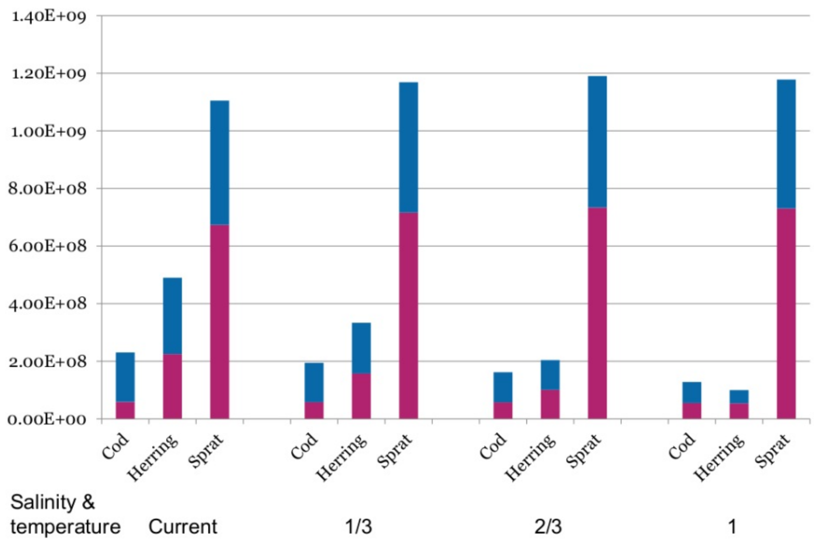

Figure 8.1.4. Adult-, and recruit productivity (in kg) of cod, herring and sprat under different environmental scenarios under current RV conditions. Adult productivity is shown as blue and the productivity of recruits is show as red. The environmental settings (of salinity and temperature) are given by the text below the bars in increments towards the full scenario of 2099 (1 indicates the full scenario of 2099).

Meier H.E.M., Eilola K., Gustafsson B.G., Kuznetsov I., Neumann T., Savchuk O.P., 2012. *Uncertainty assessment of projected ecological quality indicators in future climate*. Oceanography No 112.

Norrström N., Casini M., Holmgren N.M.A., 2016, Nash equilibrium can resolve conflicting maximum sustainable yields in multi-species fisheries management, ICES Journal of Marine Science, doi:10.1093/icesjms/fsw148

9 Response to requests to WGSAM

No requests were received prior to the 2016 meeting.

Annex 1: List of participants

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Annex 2: Agenda

ICES Working Group on Multispecies Assessment Methods (WGSAM),

Reykjavik, Iceland, 10-14 October 2016

Date	What and Who
Monday	10.00 Meeting start, welcome and housekeeping 10.30 ToR A (updates on progress), maximum 10 minute presentations per topic. Each presenter to write short report text after presentation 12:00 Lunch 13:00 ToR A continued ToR C (skill assessments), ToR D (multimodel assembly) – these are not getting focus this year, but short presentations/discussion are fine 18:00 End
Tuesday	This day will be devoted to in-depth reviews. 09:00 Presentation of in-depth reviews: • Baltic Ecopath/Ecosim Key Run, present results and assign subgroup (ToR B) • Review of LeMans time-dependant modelling framework – towards an ICES/WGSAM community model ensemble, present overview and assign subgroup (ToR D) 10:00 Meeting splits into two subgroups to work on the reviews 12:00 Lunch 13:00 Continue reviews 16:00 Key run subgroup(s) reports back in plenary 18:00 End (possibly later as required by subgroup work)
Wednesday	09:00 ToR E Performance and sensitivity of Ecosystem indicators ▪ Indicators in the North and Celtic Seas (Robert Thorpe) ▪ Indicator thresholds & performance, US (Sarah G, for Jamie Tam) ▪ A novel biodiversity indicator (Valerio Bartelino, Barbara Bauer) ▪ Testing indices of community status using a Species-Area-Relationship (SAR) model approach in relation to LFI (Large Fish Index) and MTL (Mean Trophic Level) on the Swedish west coast (Andrea Belgrano) ▪ Sensitivity of MS-MSY reference points to salinity and temperature in the Baltic Sea (Noél Holmgren, Niclas Norrström) 11:00 (Move to other room) 12:00 Lunch

	13:00	Tour
Thursday	09:00	ToR F Meta analysis on impacts of top predators on fish stocks in ICES waters ▪ Northeast US mammal/fishery simulations (Sarah G, for Laurel Smith) ▪ *Can we work on a framework for a comparison paper here?*
	12:00	Lunch
	13:00	ToR G Multispecies, mixed fishery, practical management advice ▪ Update on MSY ranges for North Sea (Robert Thorpe) ▪ Multispecies/mixed fishery advice, US (Sarah G and Sean Lucey) ▪ MS-MSY reference points for the Baltic Sea, how can they be implemented? (Noél Holmgren, Niclas Norrström)
	14:30	Move to other room
		Continue ToRs as required, writing
		Possible further reporting back from sub groups if required
	18:00	End
Friday	09:00	Continue ToRs as required, writing
	13:00	Meeting closes in order to allow people to reach 16.00 and 17.00 international flights.

Annex 3: Baltic Sea EwE model Key Run

Report on Key Run for the Baltic Sea Ecopath with Ecosim Ecosystem Model, 2004-2013

Key run summary sheet

AREA	BALTIC SEA
Modelling approach	Ecopath with Ecosim
Type of model	Foodweb compartment
Run year	2016
Species/Groups	22 functional groups
Time range	2004-2013
Time-step	Yearly (internal multistanza calculations monthly)
Area structure	Model approximately covers Baltic Proper ICES Subdivisions 25-29, excl. 28-1. A spatial extension of the model is under development.
Stomach data	From the EU tender "Study on stomach content of fish to support the assessment of good environmental status of marine foodwebs and the prediction of MSY after stock restoration"
Purpose of key run	Description of changes in the Baltic Sea foodweb
Model changes since last key run	First key run

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About this annex

This annex describes the parameterisation of the Baltic Sea Ecopath with Ecosim model key run and its calibration to time-series data 2004–2013. The contents have been presented and reviewed at the ICES WG Multispecies Assessment Model in October 2016. The report and output data files are made available *via* the WGSAM webpage. The contents Table indicates key associated files.

SECTION	ASSOCIATED SUPPORT FILES
1 The Baltic Sea Ecopath model 1.1 Functional groups 1.2 Diet composition 1.3 Multi-stanza representation of life stages 1.4 Fishing fleet structure and parameterisation 1.5 PreBal diagnostics	Basic estimates, diet composition, fleet parameters, landings, discards, off-vessel price: Baltic 2004_Key Run_Ecopath.xlsx Mortality rates, consumption, respiration, PreBal predator-prey ratios, ecosystem state indicators: Baltic 2004_Key Run_Ecopath Output.xlsx
2 Ecosim fit to time-series data 2.1 Approach overview/ strategy 2.2 Time-series data 2.2.1 Forcing time-series 2.2.2 Reference time-series 2.3 Time-series fitting using different setup specifications 2.3.1 Evaluating the goodness-of-fit 2.3.2 Time-series weighting 2.3.3 Fitting diagnostics and performance 2.3.4 Stock recruitment and MSY 2.3.5 Stability tests	'Standard' set of vulnerabilities and forcing and reference time-series as they are imported into Ecosim: Baltic 2004_Key Run_Ecosim.xlsx
3 Key run specification and setup	
4 Key Run Outputs 4.1 Model fits to data 4.2 Cod diet 4.3 Mortality 4.4 Equilibrium estimates of F 4.5 Mortality rates time-series – predation and fishery (partial F's) 4.6 Ecosystem indicator trends	biomass and catch model outputs, their confidence range and the corresponding data as presented on the relative biomass and catch plots and ecosystem indicators: Baltic 2004_Key Run_Ecosim Output.xlsx
5 References	

1. The Baltic Sea Ecopath model

Below we include a description of the Ecopath model and parameters.

1.1 Functional groups

The key run model was a further development of the model described by Niiranen *et al.* (2013) and Tomczak *et al.* (2012). In contrast to the model of Tomczak *et al.* (2012), the key run model was parameterised according to post-regime shift conditions. The functional groups included represent the most important groups in the offshore central Baltic Sea.

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

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

Grey seals and fish eating birds represent top predators. Seals are the top trophic level of the Baltic food-web and play a potential role in top-down control of fish populations. Due to significant increase of seal abundance in the Baltic (Härkönen *et al.*, 2013) and rising conflict with fisheries it was important to include this group in the model. As for fish-feeding birds, only birds categorised as offshore pelagic fish feeders by HELCOM (2012) were included: razorbill *Alca torda*, common guillemot *Uria aalge* and black guillemot *Cepphus grylle*. Fish eating birds were previously neglected in the open Baltic food-web models (Harvey *et al.*, 2003; Tomczak *et al.*, 2012). We decided to include them in order to better reflect their effects on fish as well as to represent an important link to the terrestrial ecosystem. Fish eating birds are also an indicator group of ecosystem health related to fish condition (Österblom *et al.*, 2007, 2006), which was another reason to include this group in the model.

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

Details of the parametrisation are described in Table 1.

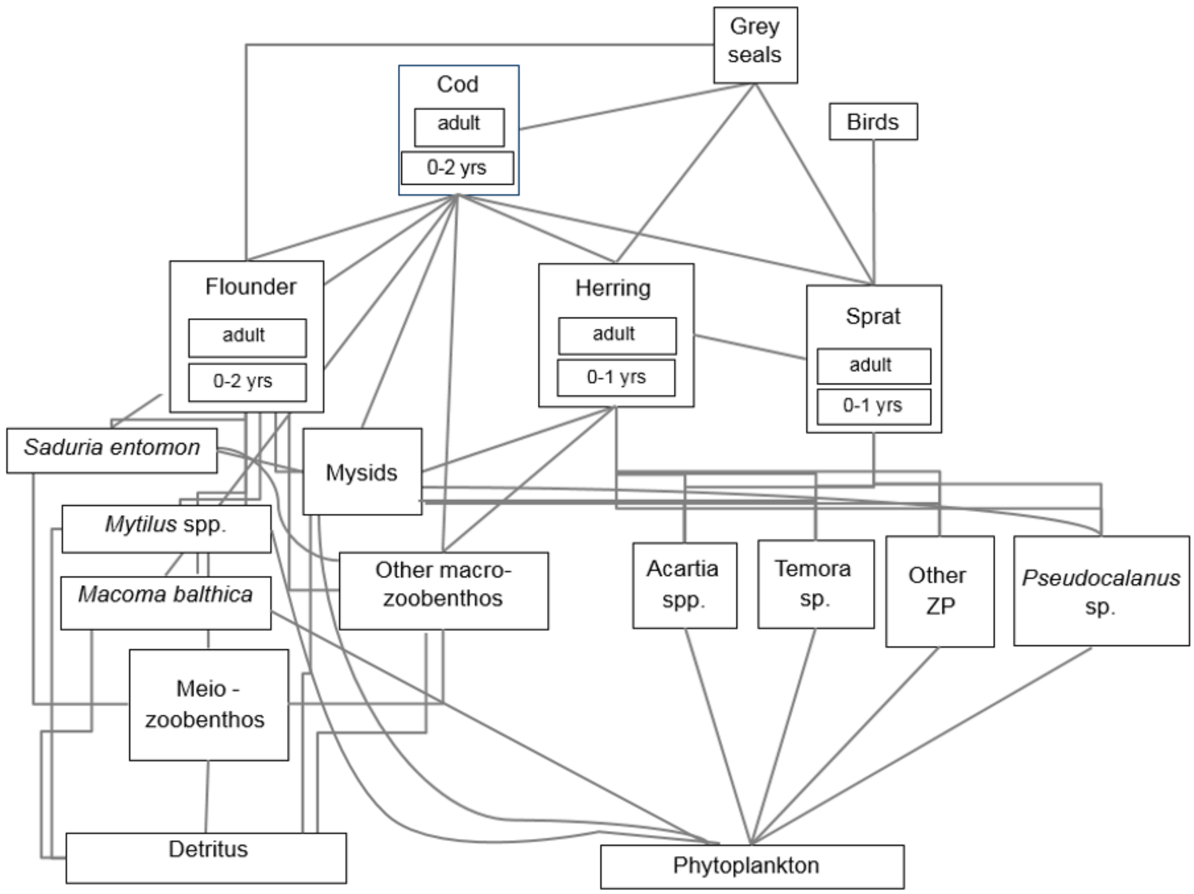


Figure 1. Functional groups and their feeding relationships in the Ecopath with Ecosim model.

Table 1. Ecopath input parameters, references and assumptions or changes implemented compared to the references, when applicable. All biomasses (B) are in units of t/km². The 'Total mortality' parameter of multistanza groups is equivalent to the P/B (production/biomass) of other groups. 'Q/B' refers to consumption/biomass, 'UA' unassimilated consumption', 'BA' to biomass accumulation rate and 'DC' to diet composition.

GROUP NAME	PARAMETER	VALUE	SOURCE	COMMENT
Grey seal	*B*	0.006	BALSAM Grey Seal	BALSAM reports estimates

			Database (cross-checked with: Lundström *et al. in press*)	on number of seals in various areas. Numbers in areas contained in the Baltic Proper were summed. Numbers were converted into density by assuming an average seal weight of 100 kg and an area of 240 000 km².
	P/B	0.1	Harvey *et al.*, 2003	
	Q/B	16.28	Gårdmark *et al.*, 2012 (cross-checked with: Lundström *et al. in press*)	Both sources report daily food consumption in kg food, calculated to yearly amount and assuming a 100 kg of average seal weight.
	BA	0.06	Härkönen *et al.*, 2013.	BA was set to zero for stability tests.
	DC		Lundström *et al. in press*	
Fish-feeding birds	*B*	0.002	Durinck *et al.*, 1994; Österblom *et al.*, 2002	Razorbill and black guillemot abundances from the early 90's were reported in Durinck *et al.*, 1994; were converted to densities assuming weights of 700 and 400 g, resp., accounting for an estimated increasing trend in the populations to the 2000's reported by Herrmann *et al.*, (2013). Common guillemot abundance estimates are by Österblom *et al.*, (2002) are similarly converted to densities, assuming an aver-

				age weight of 1 kg.
	P/B	0.1	Harris *et al.*, 2000; Lavers *et al.*, 2009	Calculated as equal to mortality (1-survival rate).
	Q/B	130	Lilliendahl and Solmundsson, 1997 Mehlum and Gabrielsen, 1993	Estimated based on daily food intake and body mass values reported for razorbill (the most abundant species in the group). Corresponding estimates for common guillemot are 170 (Enstipp *et al.*, 2006) and for black guillemot 223 (Mehlum and Gabrielsen, 1993), the latter likely being an overestimation as it is based on food intake values during the chick-rearing period.
Adult cod	*B*	0.33	ICES, 2013	Value between SSB and Age3+ biomass from SAM model output for Eastern Baltic cod (SDs 25-29, excl. Gulf of Riga). Area used for calculating density is 240 000 km^2.
	Total mortality	0.885	ICES, 2013; FishBase	Total mortality calculated as the sum of natural mortality M (0.18, FishBase, value from Gdansk Deep) and fishing mortality in 2004 calculated as: (landings+discards)/B.
	Q/B	3.81	Witek, 1995	

	UA	0.17	Harvey *et al.*, 2003	
	DC		Huwer *et al.*, 2014; ICES, 2016	Average diet 2003-2005 of cod≥33 cm, stomachs collected in SD 25-29. Also see note below.
Juvenile cod	*B*	0.354	calculated by EwE	
	Total mortality	1.062		Total mortality assumed to be 1.2 times adult total mortality.
	Q/B	7.65	calculated by EwE	
	DC		Huwer *et al.*, 2014; ICES, 2016	Average diet 2003-2005 of cod<33 cm, stomachs collected in SD 25-26. Also see note below.
Adult herring	*B*	2.33	ICES, 2015	XSA assessment of Central Baltic herring, SDs 25-29 and 32, excl. Gulf of Riga) , Age 2+. Area used for calculating density is 280 000 km^2.
	Total mortality	0.78	ICES, 2015; FishBase	Calculated as the sum of natural mortality M (0.65, FishBase, estimated using life-history tool) and fishing mortality in 2004 calculated as landings/B.
	Q/B	3	Witek, 1995	Witek, 1995 reported 1.96, adjusted to have more realistic production/consumption values.

	DC		Casini & Cardinale, 2004; Möllmann *et al.*, 2004; Tomczak *et al.*, 2012	DC values chosen within the range of DCs reported to satisfy the mass-balance assumption.
Juvenile herring	*B*	1.003	calculated by EwE	
	Total mortality	1.176		Assumed to be 1.5 times adult total mortality.
	Q/B	5.811	calculated by EwE	
	DC		Casini & Cardinale, 2004; Möllmann *et al.*, 2004; Tomczak *et al.*, 2012	DC values chosen within the range of DCs reported to satisfy the mass balance assumption.
Adult sprat	*B*	2.39	ICES, 2015	XSA assessment of sprat in Subdivisions 22-32, Age 2+. Area used for calculating density is 500 000 km^2.
	Total mortality	1.24	ICES, 2015; FishBase	Calculated as the sum of natural mortality M (0.65, FishBase, estimated using life-history tool) and fishing mortality calculated in 2004 as landings/B.
	Q/B	4.63	Witek, 1995	
	DC		Casini & Cardinale, 2004; Möllmann *et al.*, 2004; Tomczak *et al.*, 2012	DC values chosen within the range of DCs reported to satisfy the mass-balance assumption.
Juvenile sprat	*Total mortality*	1.865		Assumed to be 1.5 times

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

	P/B	1.3	Witek 1995	
	Q/B	5	Witek 1995	Changed from 6.51 during Prebal procedure
	DC		Englund *et al.* 2008	
Mytilus spp.	*B*	10	NMFRI- outer gdansk basin, mean 2002-2003 samples, Darr *et al.* 2014	
	P/B	1.75	Witek 1995	
	Q/B	8.73	Witek 1995	
	DC		Mackinson&Daskalov 2007	Based on diet of suspension feeders in the North Sea Ecopath model.
Macoma balthica	*B*	45	NMFRI- outer gdansk basin, mean 2002-2003 samples., Darr *et al.* 2014, Timmerman *et al.* 2012	
	P/B	0.4	Witek 1995	
	Q/B	2	Witek 1995	
	DC		Timmermann *et al.*, 2012	A mixture of suspension feeding (see DC *Mytilus* spp.) and deposit (detritus) feeding.
Oth. macrozoobentos	*B*	11.385	NMFRI- outer gdansk basin, mean 2002-2003 samples	

	P/B	2	Witek 1995	Assuming that most abundant groups are *Pontoporeia f.* and Polychaetes (e.g. *Harmothoe sarsi*).
	Q/B	10	Witek 1995	Assuming that most abundant groups are *Pontoporeia f.* and Polychaetes (e.g. *Harmothoe sarsi*).
	DC			A mixture of suspension feeding (bivalves, see DC *Mytilus* spp.), deposit feeding (amphipods), deposit feeding and predation (polychaetes).
Meiobenthos	*B*	6.8	Olafsson&Elmgren 1997	Summer value, conversion factor from shell-free dry weight to wet weight 1:4.
	P/B	6.17	Harvey *et al.* 2003	
	Q/B	31.17	Harvey *et al.* 2003	
	DC		Olafsson *et al.* 1999	
Mysids	*B*	2.16	estimated by EwE	Assuming an ecotrophic efficiency of 0.75 (Niiranen *et al.* 2013).
	P/B	5	Mohammadian *et al.* 1997 in Tomczak *et al.* 2012	Set to lower value than in source (7.3) to have production/respiration ratio<1 and reflect species shift (Ogonowski *et al.* 2013) from *Mysis* spp. to *Neomysis* with

				lower P/B (Witek 1995).
	Q/B	15	Harvey *et al.* 2003	
	DC		Tomczak *et al.* 2012	
Other zooplankton	*B*	4	average NMFRI 2003-2005	
	P/B	20	Niiranen *et al.*, 2013	
	Q/B	100	Tomczak *et al.* 2012	
	DC			Phytoplankton feeding
Pseudocalanus spp.	*B*	1.93	Average of: 1. BIOR 2. average NMFRI 2003-2005	BIOR data converted to density assuming an average depth of 62 m, average 2003-2005, average spring-summer across Gotland Sea and Bornholm Basin.
	P/B	7	Niiranen *et al.*, 2013; Witek, 1995	
	Q/B	27	Witek 1995	
	DC			Phytoplankton feeding
Acartia spp.	*B*	3.027	Average of: 1. BIOR 2. average NMFRI 2003-2005	BIOR data converted to density assuming an average depth of 62 m, average 2003-2005, average spring-summer across Gotland Sea and Bornholm Basin.

	P/B	20	Niiranen *et al.*, 2013	
	Q/B	83	Witek 1995	
	DC			Phytoplankton feeding.
Temora spp.	*B*	2.271	Average of: 1. BIOR 2. average NMFRI 2003-2005	BIOR data converted to density assuming an average depth of 62 m, average 2003-2005, average spring-summer across Gotland Sea and Bornholm Basin.
	P/B	20	Niiranen *et al.*, 2013	
	Q/B	83	Witek 1995	
	DC			Phytoplankton feeding.
phytoplankton	*B*	7.05	NMFRI, 2004.	Average value of open water stations (SD 25-26).
	P/B	200	BALTSEM model output, Johannson *et al.* 2004, Tomczak *et al.* 2012;	P/B from BALTSEM (Baltic Sea Long-term Eutrophication Model, Baltic Nest Institute) output was calculated as total annual production Gotland Sea (GS) and Bornholm Basin (BN) to total phytoplankton biomass standing stock GS and BN.
Detritus	*B*	1645	E. Gustafsson pers.comm	Based on BALTSEM model output, the sum of detrital POC, DOC (phytoplankton exudates, zooplankton excretion etc.) and benthic (sediment) OC (the largest pool).

	Assuming a conversion factor from C to wet weight 11.62 (Tomczak *et al.* 2012).

1.2 Diet composition

The full diet composition matrix is available in data file Baltic 2004_Key Run_Ecopath.xlsx: Diet composition. Here we only discuss the more highly resolved diet composition of higher trophic level groups (for references see Table 1 above).

Fish-feeding birds almost exclusively consumed sprat, while grey seals fed on sprat, herring and to a lesser extent on demersal fish. A significant part of grey seal diet in the Baltic consists of fish not represented in the model (import), e.g. whitefish, salmon and eels. Both clupeids are mostly zooplanktivores, but mysids and other macrobenthos are also part of herring diet, especially for adults. Adult flounder consumes mostly mussels, while juvenile flounder and *Saduria entomon* predate mostly on mysids and macrobenthos (e.g. oligochaetes, benthic amphipods).

To parametrize cod diet composition (see Figure 21, Figure 22), we used stomach data described in Huwer *et al.* (2014) and ICES (2016), collected 2003-2005. The number of stomachs sampled during these years was relatively low and they mostly came from coastal areas in SDs 26 and 28. This may explain the discrepancies between diet proportions found in these stomachs and otherwise described in the literature. To have a more robust estimate of cod diet, the stomach data was adjusted during the calibration process. The proportion of adult sprat in cod diet in general was increased as well as the proportion of adult vs. juvenile sprat in cod diet was increased (i.e., diet proportion shifted from juvenile to adult sprat) so that predation mortality by cod on sprat was more consistent with estimates from MSVPA/SMS (Figure 23) and sprat weight contribution to cod diet was consistent with analysis of cod stomachs from a longer time period. Increased proportion of sprat in the diet was compensated by a decrease in cannibalism, *Saduria entomon* and adult herring (where using proportions calculated from stomach data would have resulted in a predation mortality inconsistently high compared to assessment models). Additionally, juvenile flounder was added to adult cod diet, *Pseudocalanus* spp. to juvenile cod diet, and a few percentages were shifted from adult herring to juvenile herring in juvenile cod diet to reflect its preference for juvenile herring.

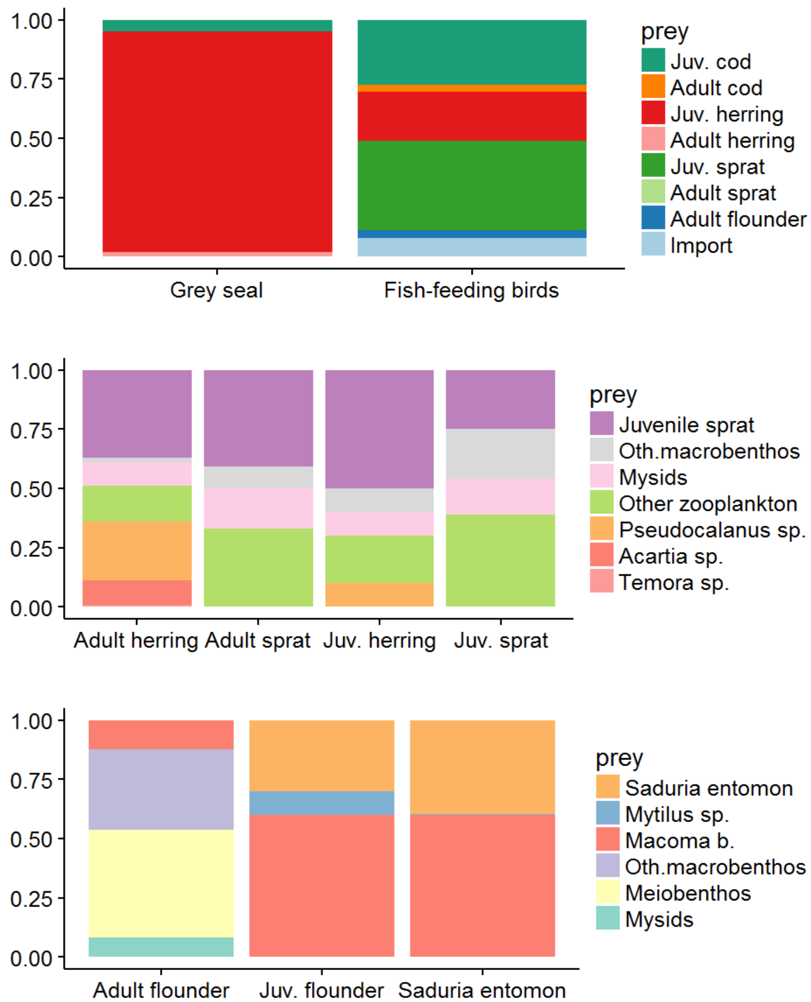


Figure 2. Diet composition of selected predator groups.

1.3 Multi-stanza representation of life stages

All modelled fish groups were represented with 2 life stages, adult and juvenile. Multi-stanza representation of life stages enables the model to account for ontogenetic changes in diet. When Ecopath includes multistanza groups, the 'usual' Ecosim differential equations to model biomass change are replaced by a set of difference equations that track monthly changes in the number and mean body weight of animals of all monthly cohorts. Production of juveniles does not occur in a yearly pulse, as in reality, but spread evenly across the annual cycle. Thus, juvenile biomass estimated by EwE models is considerably lower (appr. 1/12th) of real biomass and can only be considered a proxy variable (ICES, 2011; Walters *et al.*, 2010).

Table 2. Multi-stanza parameters based on reported values for the Baltic Sea in FishBase and literature values.

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

1.4 Fishing fleet structure and parameterisation

The model contains 10 fleets which are based on aggregation of data from different gears and size categories used in STECF (Scientific, Technical and Economic Committee for Fisheries) reports.

Table 3. Ecopath model fleet groups and their corresponding equivalents in STECF data

Ecopath fleet	STECF gears	STECF size categories
ACT0018	'DEM_SEINE','OTTER','R-DEM_SEINE','R-OTTER'	'O10T12M','O8T10M','U8M','O12T18M','U10M'
ACT1824	'DEM_SEINE','OTTER','R-DEM_SEINE','R-OTTER'	'O18T24M'
ACT2440	'DEM_SEINE','OTTER', 'R-DEM_SEINE','R-OTTER'	'O24T40M'
PAS0012	'GILL','POTS','R-GILL','LONGLINE','R-LONGLINE','TRAMMEL','R-TRAMMEL'	'O10T12M','O8T10M','U8M','U10M'
PAS1218	'GILL','POTS','R-GILL','LONGLINE','R-LONGLINE','TRAMMEL','R-TRAMMEL'	'O12T18M'
PAS1840	'GILL','POTS','R-GILL','LONGLINE','R-LONGLINE','TRAMMEL','R-TRAMMEL'	'O18T24M', 'O24T40M'
PEL0018	'PEL_SEINE','PEL_TRAWL','R-PEL_TRAWL'	'O10T12M','O8T10M','U8M','O12T18M','U10M'

PEL1824	'PEL_SEINE','PEL_TRAWL','R-PEL_TRAWL'	'O18T24M'
PEL2440	'PEL_SEINE','PEL_TRAWL','R-PEL_TRAWL'	'O24T40M'
PEL40OO	'PEL_SEINE','PEL_TRAWL','R-PEL_TRAWL'	'O40M'

The proportion of the landings and discards of each stanza was set based on data from STECF from 2004. The total amount of landings per species were set based on ICES data, as not all countries reported to STECF in 2004 and their data before 2008 is considered somewhat unreliable. Prices and relative costs of fishing are based on STECF economic data from SD 25-32 (for Poland 24-32), yearly average 2008-2010. For clupeids we assumed the same price for adults and juveniles, for cod we calculate juvenile price as 2/3 of adult price.

1.5 PreBal diagnostics

PreBal diagnostics described by Link (2010) are a way to judge the quality of Ecopath models. Below we list PreBal criteria and our model's performance on each.

CRITERIA	CORRESPONDING MODEL RESULTS	COMMENT
Biomasses should span *5-7* orders of magnitude	4.35	Slightly lower, but modelled area is not a large marine ecosystem.
Slope (on log scale) around *5-10%* decline with increasing TL.	8.7%	
Taxa notably above or below slope?	see Figure 3	Fish-feeding birds, grey seal below slope: anthropogenic impacts. *Macoma b.* above slope: this is a mostly deposit feeding organism controlled by environmental conditions and less by food web interactions.
Compared across taxa, ratio between predator and prey biomass should be <1, and ~ 1-2 decimal places, depending on TL (exception: zooplankton/phytoplankton)	see data file Baltic 2004_Key Run_Ecopath.xlsx: PreBal predator-prey ratios	'Too many' zeroes appear for rare top predators only- their biomasses are too low from the energetic perspective, as indicated above. Sprat to zooplankton ratio (therefore pelagic to zooplankton ratio) high: see discussion in the main text of the report.
Q/B ratios for adult fish should be 2-4.	see data file Baltic 2004_Key Run_Ecopath.xlsx: Basic estimates	
Generally decreasing vital rates with increasing trophic level (exc. homeotherms).	see Figure 4	Sessile organisms (two mussel species) below slope-line and small zooplankton above.

Compared across taxa, ratio between predator and prey vital rates should be <1, and ~ 1-2 decimal places, depending on TL (exception: homeotherms, zooplankton-phytoplankton; for detritivores cannot be calculated)	see data file Baltic 2004_Key Run_Ecopath Output.xlsx: PreBal predator-prey ratios
No taxa have P/B higher than primary producers.	see Figure 4 left panel
P/Q < 1 for all groups	see data file Baltic 2004_Key Run_Ecopath.xlsx: Basic parameters
P/R < 1 for all groups	see data file Baltic 2004_Key Run_Ecopath Output.xlsx: Respiration
Total production and consumption should decrease with increasing trophic level.	see Figure 5
EE < 1 for all taxa	see data file Baltic 2004_Key Run_Ecopath.xlsx: Basic estimates

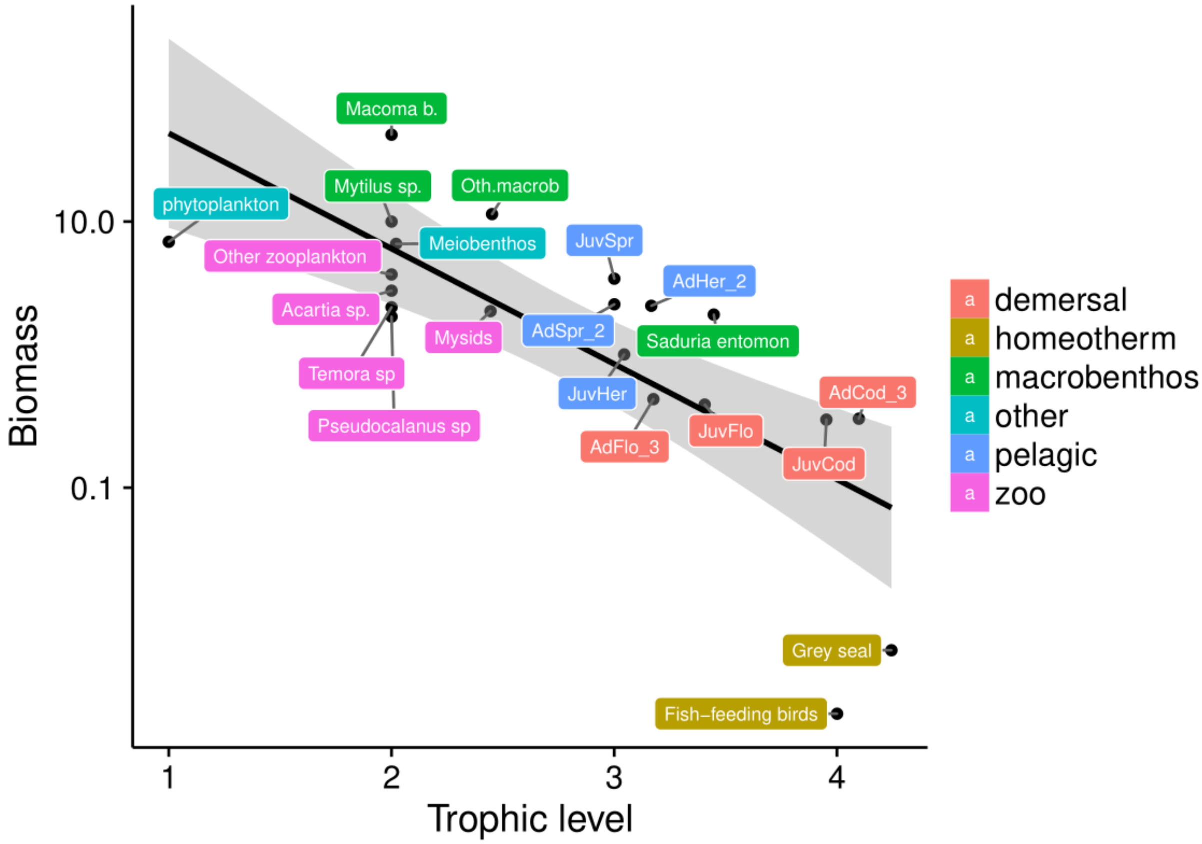


Figure 3. Declining biomass with increasing trophic level. Line: linear regression biomass~TL. Colours represent groupings used in further PreBal analysis.

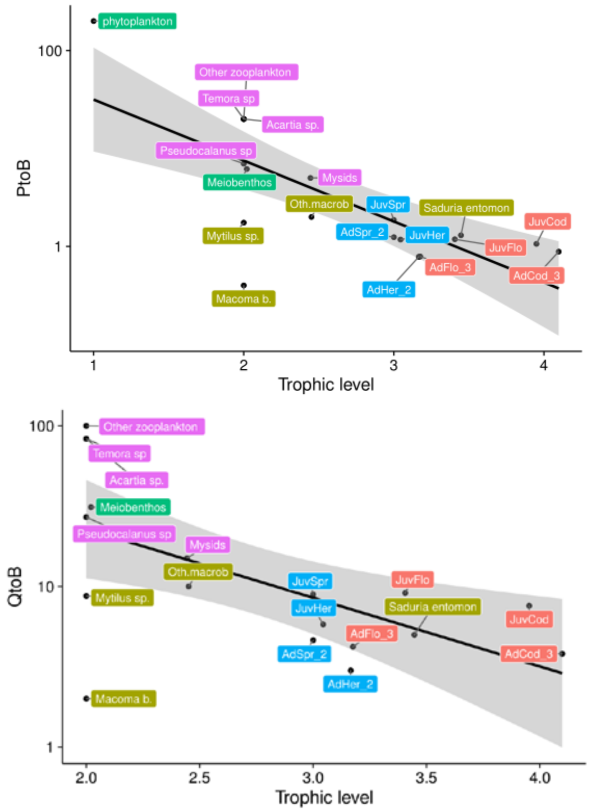


Figure 4. Declining P/B (left panel) and Q/B (right) ratios with increasing trophic level. Line: linear regression vital rate~TL. For the interpretation of colors see Figure 3.

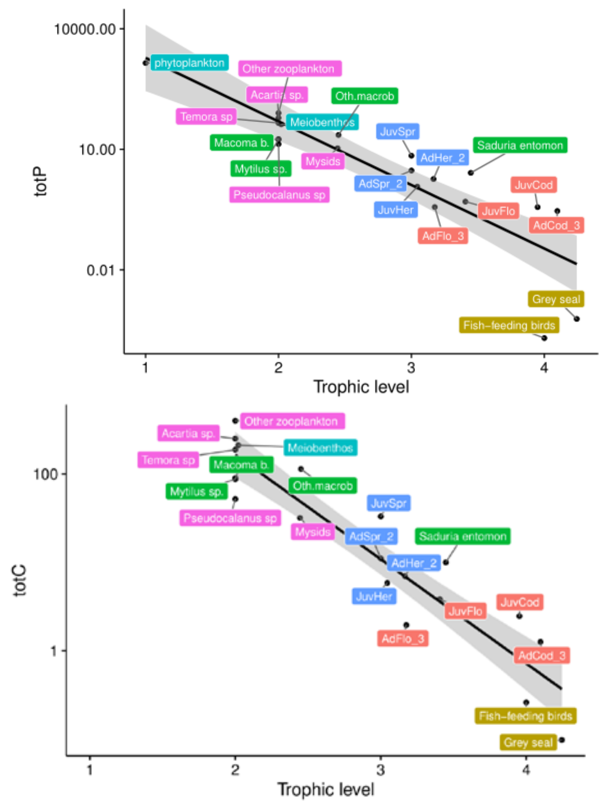


Figure 5. Declining total production (left panel) and consumption (right) with increasing trophic level. Line: linear regression vital rate~TL. For the interpretation of colors see Figure 3.

2. Ecosim fit to time-series data

2.1 Approach overview/ strategy

The Ecosim model has the balanced Ecopath as initial state and models how the ecosystem changes compared to that state as an effect of forcing. There are a few additional parameters required by Ecosim compared to Ecopath. 'Feeding time adjustment rates' were set to 0.5 for vertebrates and 0 for all other groups. 'Fraction of other mortality sensitive to changes in feeding time' was set to 0 for seals, 0.5 for birds and adult cod and 1 for all other groups (see main text for further details on the choice of these parameters).

The model selection procedure for Ecosim models involves finding values for the so-called 'vulnerability' parameters that result in credible model behavior (in fact, in the latest versions of EwE these are not the vulnerability parameters, v_{ij}, commonly used in equations when describing Ecosim processes, but multipliers for v_{ij}). The value of these parameters (noted as v below) indicates how much a predator can maximally increase its feeding on prey when its own biomass increases compared to Ecopath values. v=1 means that the maximum consumption/unit biomass prey cannot increase higher than the Ecopath level. v=2 is the default setting. The effect of increasing v on dynamics diminishes at high values of v. Small values of v are traditionally interpreted as 'bottom-up control', values around 2 'mixed control' and high values as 'top-down control'.

The strategy of our model fitting was:

1. Weighting of reference time-series (Section 2.2.2).

2. Determining the number of different v-s to fit using the 'Stepwise fitting tool' (Scott *et al.*, 2016; Section 2.3.1). This indicated 14 v-s to fit.

3. Using the 'Fit to time-series' tool in Ecosim to determine the 14 v-s the model fit is most sensitive to and estimate their values.

4. Evaluating the credibility of model behavior based on the following criteria: fit to historical data (Section 2.3), stock-recruitment and F-catch relationships (Section 2.3.4), model stability (Section 2.3.5).

5. Adjust v values for juvenile and adult sprat to improve F-catch estimates (for details see Section 2.3.1)

2.2 Time-series data

All time-series data, as imported to Ecosim, is available in the spreadsheet 'EcosimForcingFitting Standard' in the file Baltic 2004_Key Run_Ecosim.xlsx, with the exception of phytoplankton and zooplankton biomass time-series as these data are not public. All original data and calculations are available upon request.

2.2.1 Forcing time-series

There were three types of forcing applied in our model: fishing effort, fishing mortalities and abiotic forcing (Table 4, Figure 6, Figure 7).

Fishing in Ecosim is represented by yearly fishing mortalities (F= catch/biomass). Fishing mortalities can be directly imported as time-series, but Ecosim can also calculate them from partial fishing mortalities in the Ecopath base year, 2004, (calculated based on fish biomass, P/B, landings and discards composition of different fleets) and a relative effort time-series. We used the combination of these two approaches. Effort forcing time-series were constructed as the relative change (compared to 2004) in kW days at sea for each fleet segment, in SDs 25-28. We excluded data from Finland and Estonia because they started reporting in 2013 and 2005 only, resp., and inclusion of their data would have introduced an artificial jump in the effort time-series. We assumed a 2% increase in efficiency per year (i.e., a 2% increase in F/effort per year). We used fishing mortalities calculated from effort for demersal fish. However, for clupeids these calculated F's were inconsistent both with assessment yield/biomass time-series (Figure 8) and with the observed trends in catches. More specifically, there was a jump in catches and assessment yield/biomass 2006–2010 for both species, which could not be explained by the effort time-series (which showed a constant decline for pelagic trawlers 24-40 m, which are the most important fleet segment catching clupeids) or by changes in biomasses as neither surveys nor assessments indicated a jump in biomasses during this period (Figure 6, Figure 20) Thus, for herring and sprat we used yield/biomass time-series based on assessments (XSA). The reasons for the discrepancy could be for example wrong allocation of amount of landings among differently sized fleet segments in 2004 due to misreporting (wrong Ecopath partial Fs), wrong effort time-series (probably due to differences among effective effort and nominal effort), within-country differences in temporal changes in the effort missed by our aggregated model, or it could arise as herring and sprat are schooling fish for which effort may not be a good indicator of fishing mortality in general.

We applied the same type of abiotic forcing functions as Niiranen *et al.*, (2013), exc. salinity forcing on Pseudocalanus spp., as this forcing had negligible effects on model performance in their study. Environmental forcing functions used in the model come from different sources (see metadata ICES, 2008) and have been compiled and used by the ICES/Helcom Working Group on Integrated assessment of the Baltic Sea (ICES, 2015b, 2008).

Table 4. Time-series forcing used in Ecosim.

Forcing series	Group(s)	Target variable	Source
Fishing effort	fishing fleets (cod, flounder)	effort	STECF database, see text
Fishing mortalities (yield/biomass)	herring, sprat	F	Herring and sprat: catch/biomass of the relevant age classes (Table 1) based on assessments (ICES, 2015a).
primary production forcing	phytoplankton	asymptote of growth equation	Model hindcast of the BAltic sea Long-Term large-Scale Eutrophication Model (BALTSEM, Gustafsson 2003): area-weighted average yearly P/B values, SD 25-29, excl. Gulf of Riga.
Spring temperature 40-60m Gotland Sea	*Acartia* spp., *Temora* spp.	Search rate	ICES/Helcom Working Group on Integrated assessment of the Baltic Sea (see metadata ICES, 2008). Data update: ICES, (2015a)
Anoxic area (reversed)	Other macrobenthos, mysids	Search rate	ICES/Helcom Working Group on Integrated assessment of the Baltic Sea (see metadata ICES, 2008). Data update: ICES, (2015a)
Cod reproductive volume	Cod	Egg production	ICES/Helcom Working Group on Integrated assessment of the Baltic Sea (see metadata ICES, 2008). Data update: ICES, (2015a)
Summer sea surface temperature, Gotland	Herring, Sprat	Egg production	ICES/Helcom Working Group on Integrated assessment of the Baltic

Basin	Sea (see metadata ICES, 2008). Data update: ICES, (2015a)

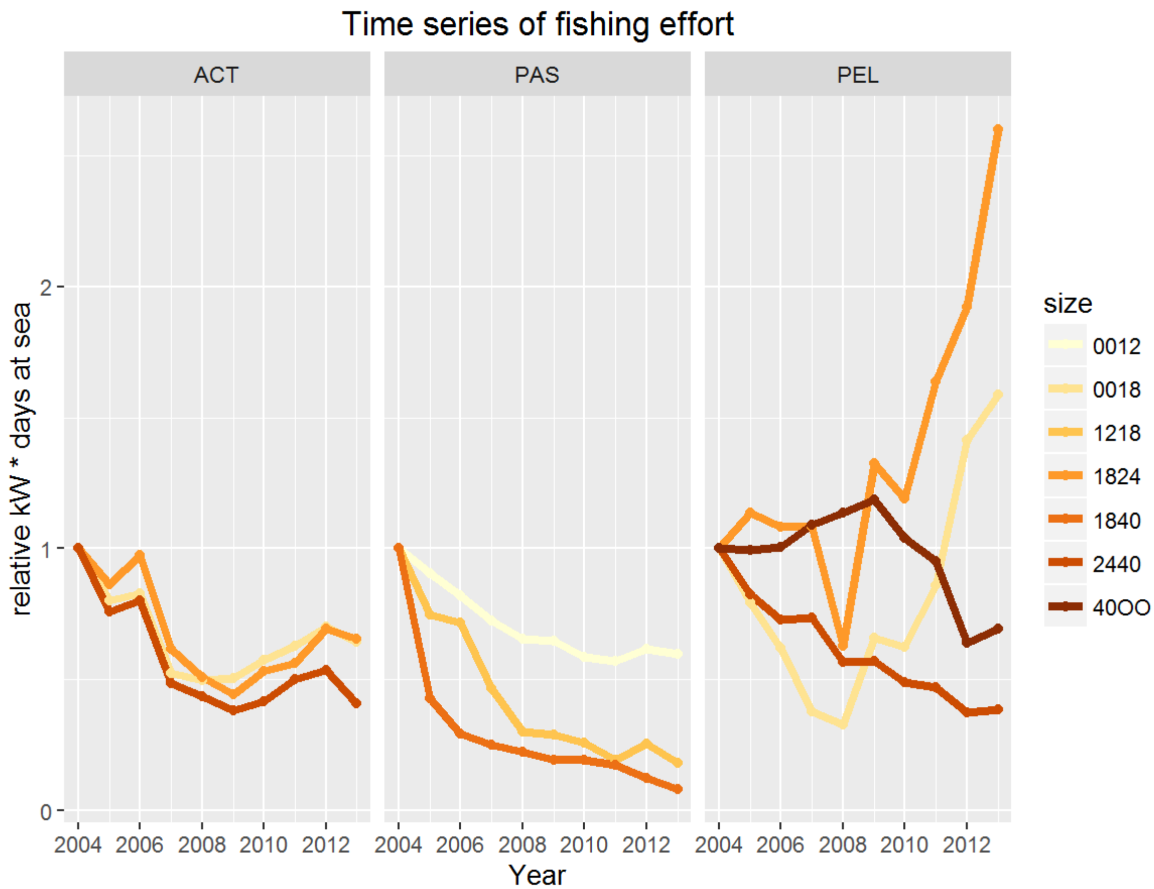


Figure 6. Time-series of fishing efforts applied, for differently sized fleet segments within three types of fleets. For abbreviations see *Table 3*.

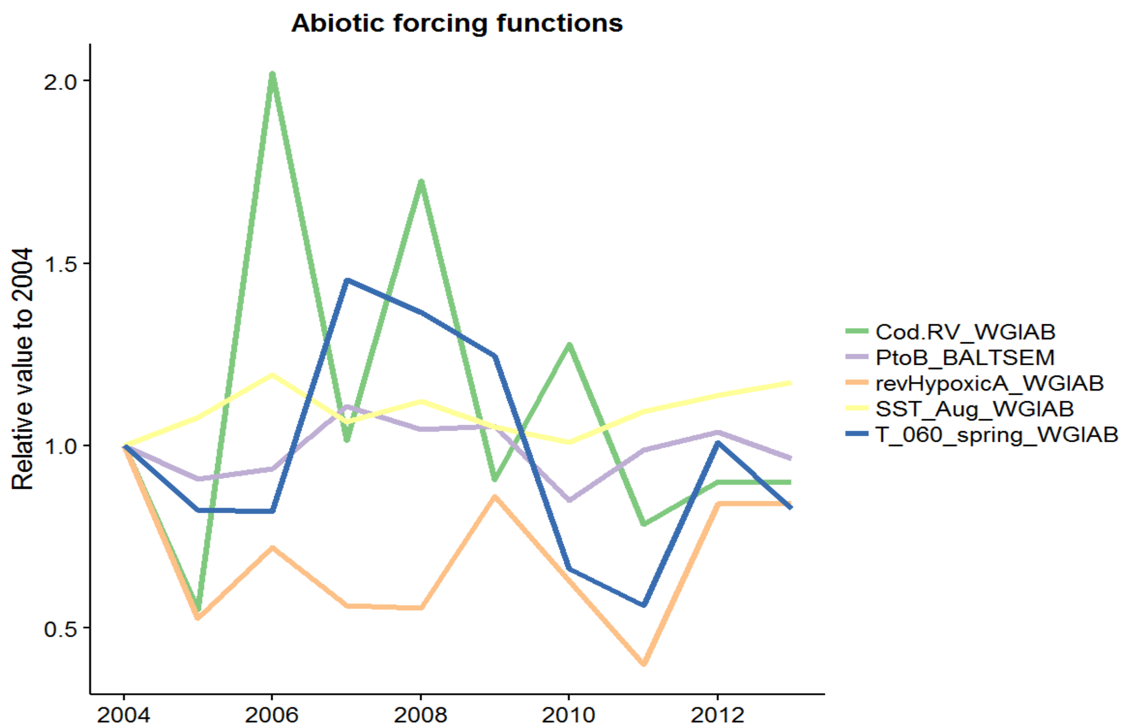


Figure 7. Abiotic forcing functions (anomalies) applied.

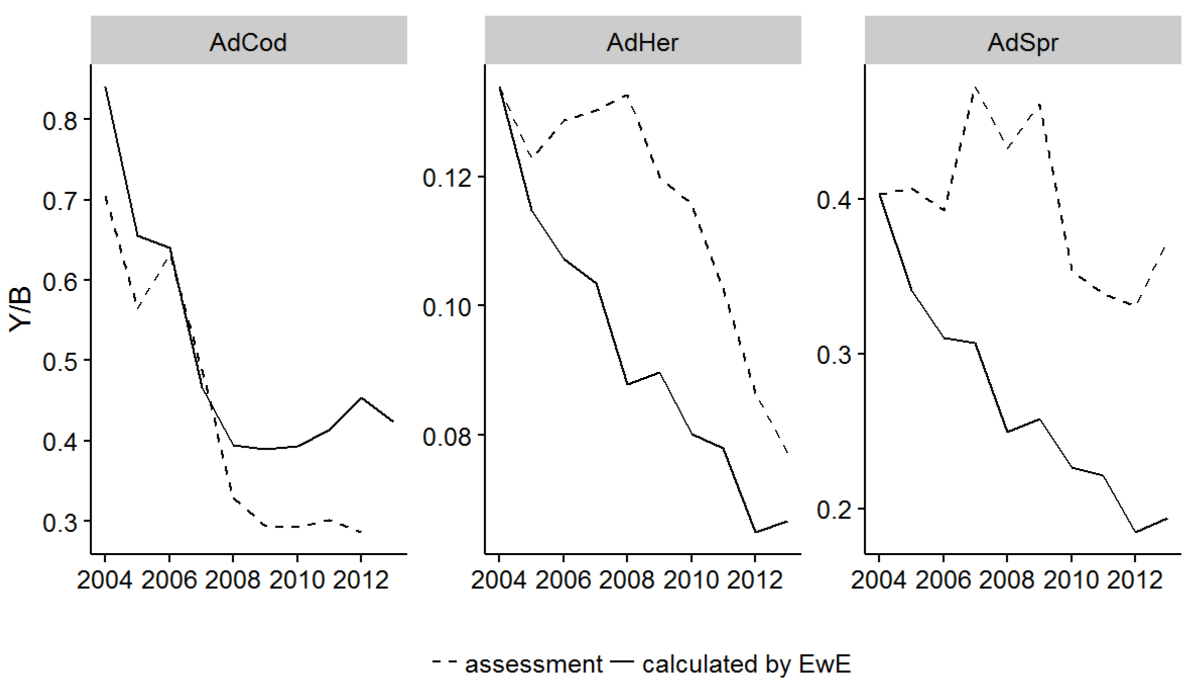


Figure 8. Fishing mortality calculated by EwE based on effort time-series only (solid lines) and yield/biomass time-series calculated from assessments based on catch and biomass time-series data processed as described in Table 5.

2.2.2 Reference time-series

The aim of using reference time-series when calibrating an Ecosim model is that the dynamics of as many as possible groups are subjected to constraints according to data. Reference time-series from 3 sources were used: survey data, assessments (relative biomasses) and catches (Table 5).

The BALSAM Grey Seal database is available as an Excel table upon submitting a request to HELCOM and it contains counts of grey seals at the HELCOM sub-basin level. As a reference data for fish functional groups a number of data sets was used depending on the species and data availability. We mainly used two types of reference data on biomasses, one based on survey indices and second on results of assessment models. Survey data can be derived from the DATRAS database, available at the ICES website. We used survey indices from the Baltic International Trawl Surveys (BITS) 1st quarter for cod and the combined survey index based on the 1st and 4th quarters for flounder, while Baltic International Acoustic Survey (BIAS) for pelagic clupeids (sprat and herring). Since the BITS does not cover the whole modelled area, in addition data from gillnets surveys for the northern flounder stock (ICES SD 27-29) was used, derived from ICES, (2016b). Assessment models for the Baltic Sea are provided by the ICES Working Group on Baltic Fish Stock Assessment. The WG uses single species models, an age structured state-space assessment model (SAM) for eastern Baltic cod and extended Survival Analysis (XSA) for sprat and herring stocks (ICES, 2015a, 2013). Annual catch data per stock for all fish species was used from WGBFAS (ICES, 2015a), as the most reliable and updated data set. One of data sources used for calibration of zooplankton functional groups (Acartia sp, Temora sp and Pseudocalanus sp) we used from ICES WGIAB data set (ICES, 2015b, 2008). WGIAB zooplankton monitoring data belong to Institute of Food Safety, Animal Health and Environment - "BIOR". That's the longest existing zooplankton monitoring data at the Baltic sea (for more details see ICES, 2015b, 2008). Zooplankton and phytoplankton 'NMFRI' time-series were provided to us within the MareFrame project (Co-creating Ecosystem-based Fisheries Management Solutions, EU FP7) and are the Polish contribution to the HELCOM COMBINE Programme. In most of the cases, samples were taken 5 times per year using the WP-2 net. All necessary calculations and processing were done using R (R Core Team, 2016), scripts available upon request to B. Bauer.

We assigned weights according to the relevance and reliability of time-series data (Christensen *et al.*, 2008) and used this combination of weights as 'Standard weighting' during model calibration (Table 5). We tested the sensitivity of fitted parameters to weights (see Section 2.3.2). For 'Standard weighting', we assigned a weight of 100 to adult fish catches, as catch data is relatively reliable and it is of high importance that the model captures dynamics of catches well. Adult biomass survey data was assigned a weight of 80, as fish biomass is also an important variable for the model to represent well, but biomass survey data is more 'noisy' than catch data. Adult biomass time-series from assessments were included as experts working on assessments usually study available data about fish stocks in such a detail that would be out of scope for our project and account for uncertainties in survey and catch data. However, we still assign a relatively small weight to these time-series as we aim to parametrize our model relatively little dependent on assessments. Juvenile fish catches were assigned a weight of 10, as catch data on juveniles is more uncertain and the juvenile groups modelled by EwE are not completely comparable to juvenile age groups in catch data. For similar reasons we assigned a weight of 1 for both types of juvenile biomass time-series. Grey seal biomass survey data was assigned a weight of 1 as modelling the dynamics of this marine mammal is not the primary focus of our model and it is represented in a simple single-group manner instead of multistanza. Zooplankton and phytoplankton survey time-series were similarly assigned a weight of 1 because of their high interannual variability due to hydrographic and physical conditions and because plankton dynamics are only modelled in a simplified manner in EwE (e.g. it doesn't include microzooplankton and has yearly time steps which are quite long compared to the generation times of these organisms).

Table 5. Reference time-series used for fitting in Ecosim, 2004–2013. The column 'Weight' describes the weights used by the Stepwise fitting plug-in and when determining the following 'standard' vulnerabilities. Sensitivity analysis on weighting is described in Section 2.3.2. For references see text above.

Group name	Time-series type	Weight	Source
Grey seal	Biomass- survey	1	BALSAM Grey seal database. Sum of counts in Baltic Proper subbasins, converted to biomass density assuming 100 kg/seal and an area of 240000.
Adult cod	Biomass- survey	80	BITS Q1 CPUE of fish >=33 cm, numbers multiplied by average weight, average of SD 25-29.
	Biomass- assessment	10	SAM Age3+, WGBFAS 2013, numbers*WEST
	Catches	100	WGBFAS, Catch in numbers (incl. Misreporting correction and discards) * WECA; Age3+
Juvenile cod	Biomass- survey	1	BITS Q1 CPUE of fish <33 cm, numbers multiplied by average weight, average of SD 25-29.
	Biomass- assessment	1	SAM Age1-2, WGBFAS 2013, numbers*WEST
	Catches	10	WGBFAS, Catch in numbers (incl. Misreporting correction and discards) * WECA; Age1-2
Adult herring	Biomass- survey	80	BIAS Q4 survey indices by age, numbers * average weight; Age2+
	Biomass- assessment	10	XSA Age2+, WGBFAS 2015, numbers*WECA
	Catches	100	landings data from WGBFAS; Age2+; Gulf of Riga excl.
Juvenile herring	Biomass- survey	1	BIAS Q4 survey indices by age, numbers * average

			weight; Age1
	Biomass- assessment	1	XSA Age1, WGBFAS 2015, numbers*WECA
	Catches	10	landings data from WGBFAS; Age1; Gulf of Riga excl.
Adult sprat	Biomass- survey	80	BIAS Q4 survey indices by age, numbers * average weight; Age2+
	Biomass- assessment	10	XSA Age2+, WGBFAS 2015, numbers*WECA
	Catches	100	landings data from WGBFAS, Age2+
Juvenile sprat	Biomass- survey	1	BIAS Q4 survey indices by age, numbers * average weight; Age1
	Biomass- assessment	1	XSA Age2+, WGBFAS 2015, numbers*WECA
	Catches	10	landings data from WGBFAS, Age1
Adult flounder	Biomass- survey	50	WGBFAS, weighted (by landings) mean of rescaled survey indices from SDs 24-25, 26-28 (fish >= 20 cm) and 27 (stations Muskö, Kvädöfjärden).
	Catches	60	WGBFAS, total landings from SDs 25-29.
Acartia spp., *Temora* spp., *Pseudocalanus* spp., Other zooplankton	Biomass-survey	1	Two time-series for each group: one based on NMFRI data (station P40/P140, located in ICES rectangle 40G86, yearly average of months 8-9), the other is from the WGIAB dataset.
Phytoplankton	Biomass-survey	1	NMFRI, average value of open water stations

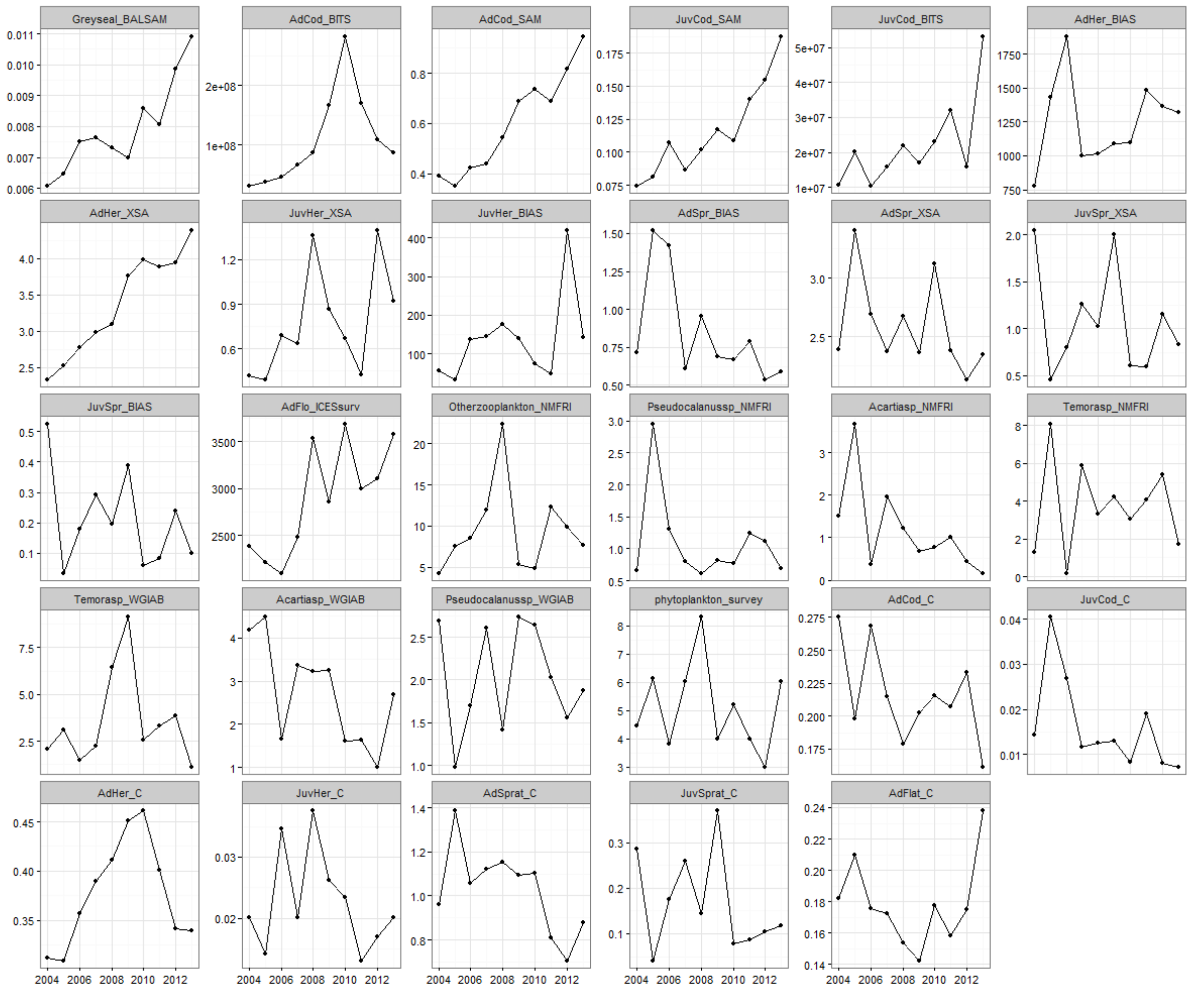


Figure 9. Biomass and catches ('_C') in units of t/km² or as survey biomass index (BITS, BIAS and ICESsurv time-series) described in Table 5. These time-series were used as relative biomass and catch series in model fitting.

2.3 Time-series fitting using different setup specifications

2.3.1 Evaluating the goodness-of-fit

Fit in Ecosim is measured as weighted sum of squared deviations (SS) of model projections from reference time-series. In case of biomass reference time-series, only relative changes are taken into account, not absolute values. Thus, it is possible to use biomass survey indices without the need for rescaling.

During the model selection process, first we assessed the sensitivity of SS to the number of 'vulnerability blocks' (v-s) fitted using the 'Stepwise fitting' plug-in of Ecosim. The plug-in iteratively fits the model, changing an increasing number of v-s compared to the default value and calculates SS in each iteration. SS decreased when fitting additional v-s until appr. 14 v-s fitted, after which fitting further parameters did not substantially improve the fit any more (Figure 10). A model fitted using 14 v-s was also indicated as the 'best model' by the lowest AIC and AICc scores among all alternatives (AIC: 342.9 AICc: 344.2).

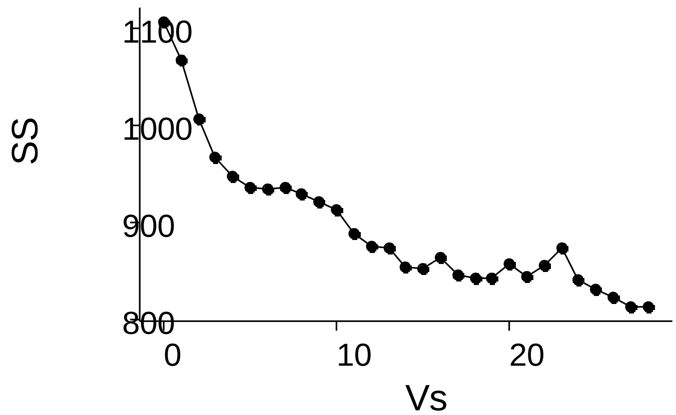


Figure 10. Change in SS as a function of number of vulnerability blocks fitted, calculated by the Stepwise fitting plug-in.

The 'Stepwise fitting' plug-in does not include forcing functions but in earlier tests we found it a good indicator for the 'best' number of *v*-s to fit. Thus, we subsequently fitted the model using the 'Fit to time-series' tool fitting 14 *v*-s using all forcing functions. Thus, we used this model version as our 'standard' parametrization, with the exception that we changed all juvenile and adult sprat *v*-s from 2 to 40 and 5, respectively. We judged this necessary as with default *v*-s for these groups the 'MSY Search' procedure produced unrealistically high catches at high F-s for sprat, as is often the case in Ecosim models when using relatively low or default *v*-s (Heymans *et al.*, 2016). As described for the 2015 North Sea key run (ICES, 2015b), we obtained an initial estimate for *v* using the formula v_init = [1-(Bunf/Bo)]/[1-(e/M)(Qo/Bo)], where, e is the growth efficiency (P/Q), Bunf is the historical max biomass, Bo is biomass in the model base year, M is the base total natural mortality rate for the predator, and Qo/Bo is the Ecopath base Q/B for the species. This gave a value of 5.06 for adult sprat and 41.65 for juvenile sprat. This change did not largely influence model fit. For final *v*-s see values in the supporting file Baltic 2004_Key Run_Ecosim: Vulnerabilities. Final SS was 858.6.

2.3.2 Time-series weighting

Time-series are weighted differently when calculating sum of squares according to our judgement on data reliability and relevance. For the exercise described in Section 2.3.1 we used weights as in Table 5 ('Standard weighting'). As the choice of weights is inevitably somewhat subjective, we repeated model fitting using the 'Fit to time-series' tool and 14 *v*-s using two different weightings besides the standard.

'Only observation': assessment data is not used as biomass reference time-series but fishing mortality forcing for clupeids is based on yield/biomass time-series from assessments. This type of fitting is the most independent of assessment models from all variations considered, but it is also the most sensitive to data uncertainties.

'Only assessment': for assessed fish groups only biomass time-series from assessment models are used as reference biomass series (otherwise same as 'Standard'). This type of fitting implicitly utilizes the knowledge of experts conducting assessments who deal with uncertainties in observation data. However, it makes the model more dependent on, and, consequently, less comparable to single-species assessments.

'Standard' weighting used both assessments and observations, but giving larger weight to surveys than assessments. More details about the rationale when assigning weights are in Section 2.2.2.

The 'type of trophic control' (see Section 2.1) in our model was relatively robust to the type of weighting we applied (Figure 11). Fitted *v* values for 13 out of 15 predator-prey pairs indicated the same type of control under all three weightings. Juvenile cod-juvenile sprat *v* changed between default and top-down control based on the setting. There was one case when the type of control changed to the opposite, adult cod predation on adult sprat. When using only assessment biomass reference time-series for fitting, v~1 while under the setups using surveys it has values ~17 and 20. The probable reason for this probably is that survey indices show a larger decline in sprat biomass than the biomass time-series from the assessment. The decline coincides with increasing cod biomass. When the fitting tool attempts to fit primarily to the survey time-series with the larger decline in sprat biomass, it chooses higher vulnerability values in the cod-sprat interaction, as these enable cod to increase its predation on sprat more strongly as its own biomass increases, thereby the model will produce a more strongly decreasing sprat biomass series, consistent with survey data. Based on historical time-series, Casini *et al.* (2008) suggested that sprat is top-down controlled by cod as opposed to herring, which is consistent with our 'Standard' parametrization. However, applying a low vulnerability value for this predator-prey pair could also make ecological sense as during the recent decades the center of distribution of sprat moved towards the Northeastern part of the Baltic Sea, while cod moved to the opposite direction. Thus, a substantial part of the sprat population is currently inaccessible for cod and not only cod biomass but also its spatial range would have to increase for it to increase its predation pressure on sprat.

To sum up, even though generally fitted *v*-s were not too sensitive to the applied weighting, this exercise highlighted the few most sensitive *v*-s, especially those related to sensitivity of sprat to cod predation. When using the model for scenario simulations, the sensitivity of model projections to the settings of at least these specific *v*-s should be investigated.

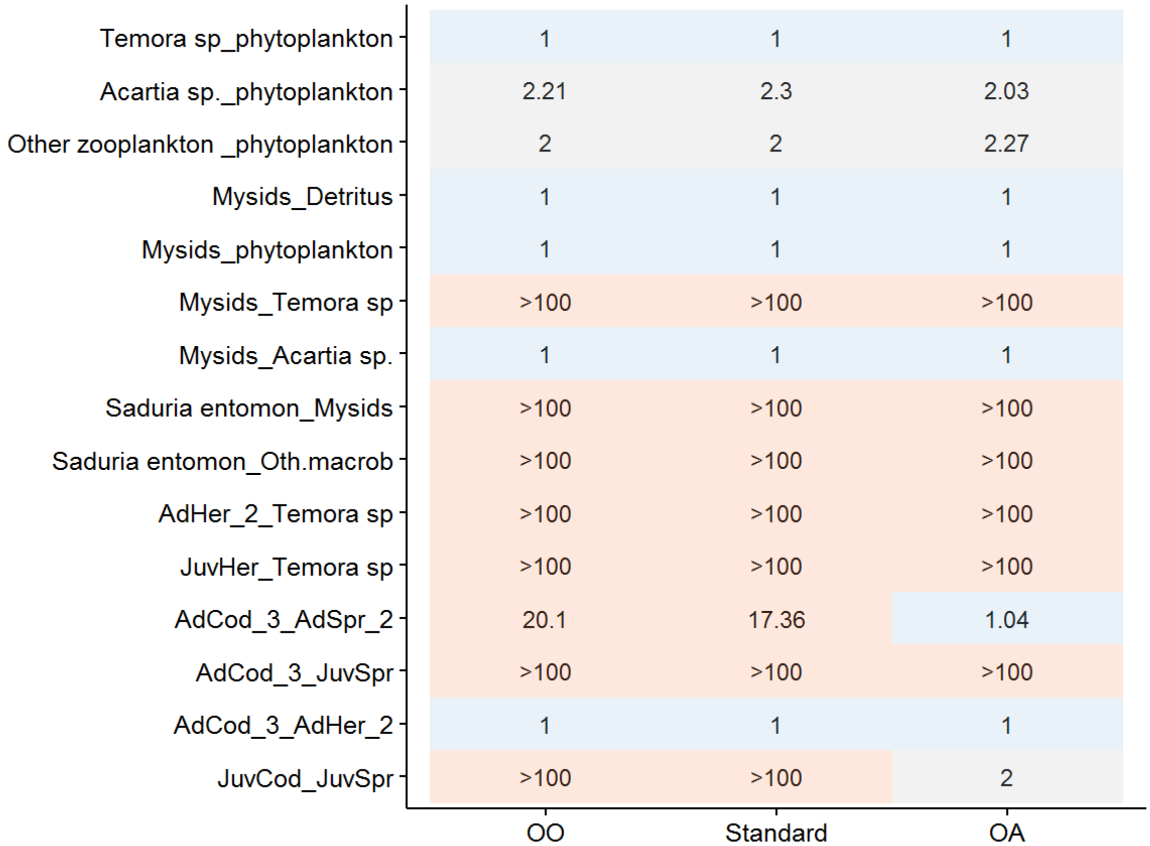


Figure 11. Fitted vulnerability values for predator-prey pairs using three different weightings of reference time-series: OO ('Only Observation'), 'Standard' and OA ('Only Assessments')- see Section 2.3.2. Cell colors reflect cell value, blue: low, gray: default, red: high. Predator-prey pairs not included in this figure had vulnerability values=2 in all setups, except adult and juvenile sprat with v=5 and 40, resp., on all their prey as described in Section 2.3.1.

2.3.3 Fitting diagnostics and performance

Plots of residuals of model predictions to observation data are given in Figure 12 and Figure 13 and plots contrasting model predictions against observation data are shown in Figure 18, Figure 19 and Figure 20. The model systematically underestimates biomass of adult clupeids compared to assessments (Figure 12), which also results in an understimation of the catches (Figure 13), as we use assessment Y/B as forcing. The opposite pattern in true for juveniles, to a larger extent in the case of sprat than herring. In the case of lower trophic levels, the model generally predicts less variable dynamics compared to that seen in observation data (Figure 12).

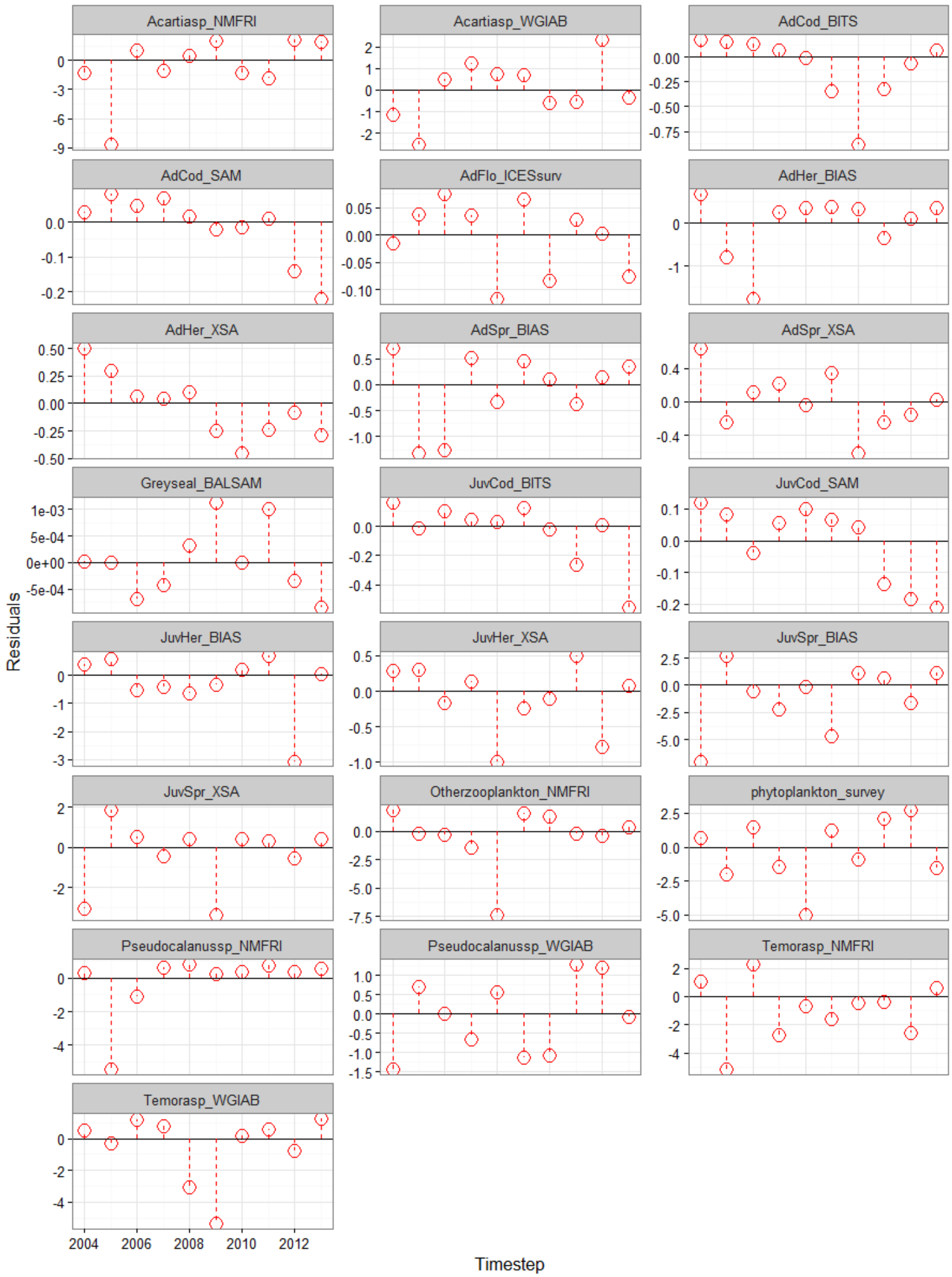


Figure 12. Residuals for model projections of relative biomasses against data.

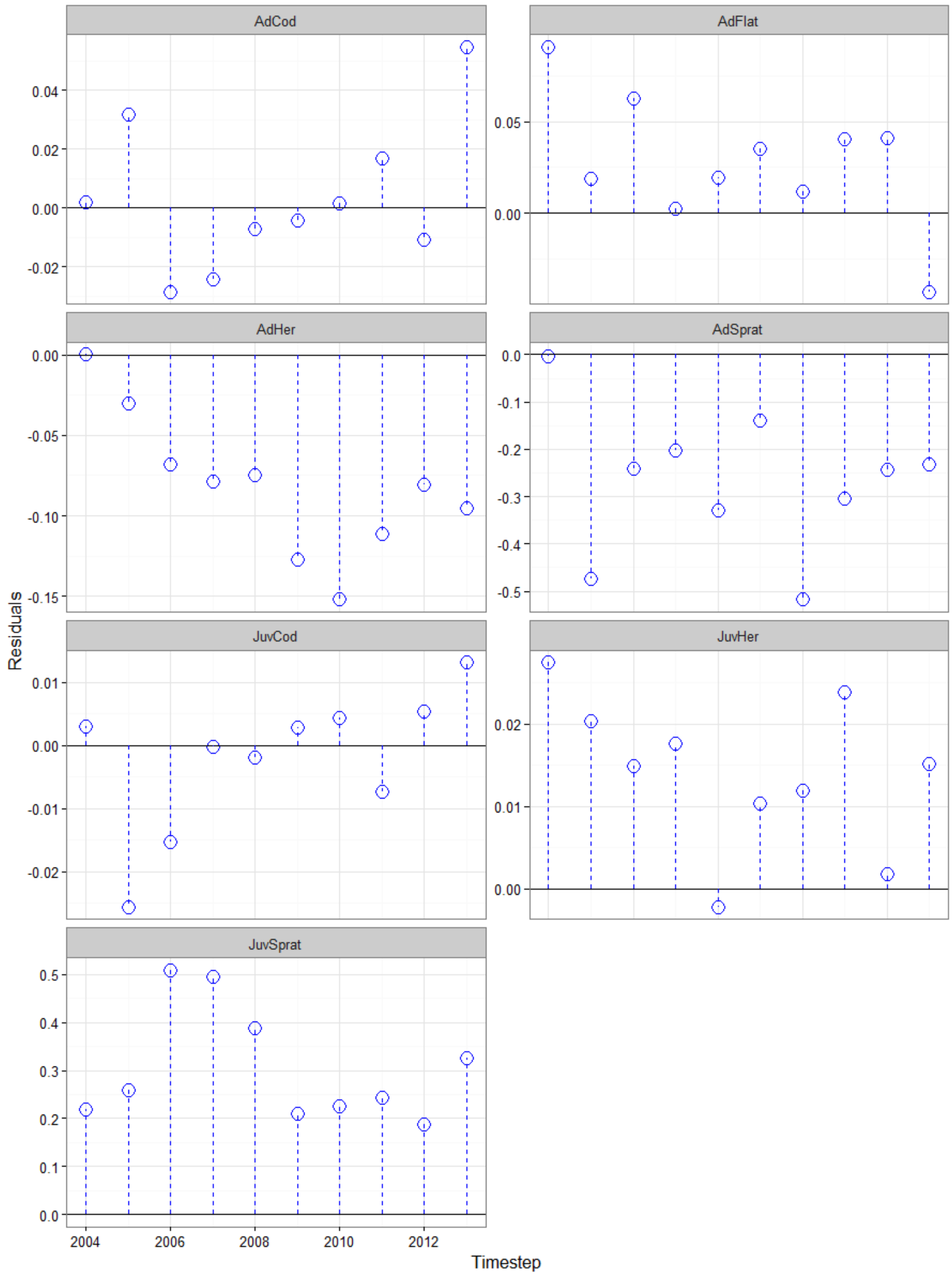


Figure 13. Residuals for catch plots.

2.3.4 Stock recruitment and MSY

We also tested the emerging stock-recruitment (S-R) relationship in the model by applying a large range of fishing mortalities simultaneously on all species to have a large range of adult biomasses (Figure 14). The emergent S-R patterns are dependent upon both the effects of the fishing pattern and the multispecies interactions that result from them. They provide an indication of how recruitment generally changes as adult biomass changes in the model. Even though this test has been used in previous key run reports, it is not completely robust, as the emergent S-R patterns are somewhat sensitive to how exactly the fishing mortality pattern is applied (i.e. how quickly the fishing mortality decreases and subsequently increases, and to which level). The key-run model generally produced dome-shaped relationships for cod, and saturating relationships for flounder and clupeids.

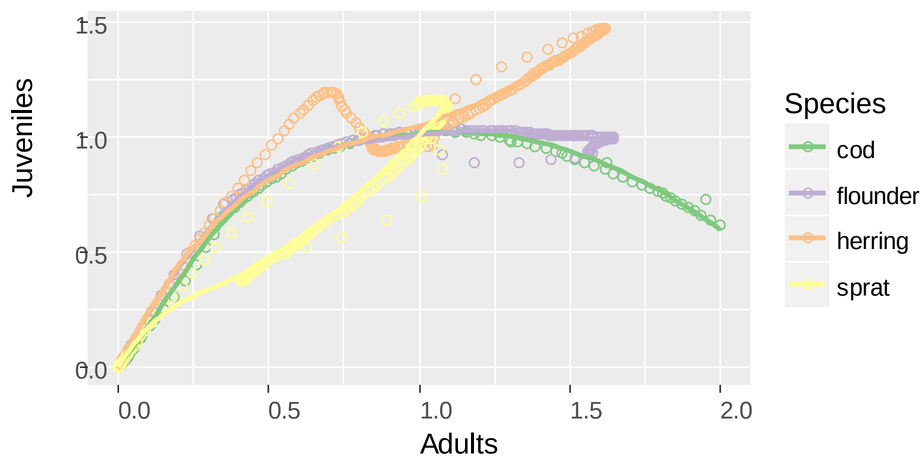


Figure 14. The relationship between number of adults and juveniles predicted by Ecosim, which can be interpreted as emergent stock recruitment relationship in the model. Each dot represents values from one model year, lines are a smoothed conditional mean added for easier visualization, generated by the geom_smooth function from the R package ggplot2. The particular run presented on the figure was done by decreasing Ecopath level fishing to 0 and subsequently increasing to appr. 20x Ecopath level, i.e. applying a V-shaped fishing pattern.

We also investigated the equilibrium relationship between F's and catch levels (Figure 15). This assessment can be performed in two ways. During the 'stationary' assessment Ecosim runs a long-term simulation at all levels of F and only the biomass of the targeted stanza reacts to changes in fishing mortality but the biomasses of all other species are kept constant. Thus, this analysis does not take into account indirect effects on the biomass of the targeted stanza via trophic linkages. In contrast, during the 'non-stationary' assessment indirect effects are taken into account to some extent. As a decrease in predator biomass at high F-s mostly results in increasing prey biomasses (i.e., more available food for the predator), full compensation assessments usually result in higher catches at higher F levels than stationary assessments. Corresponding F_{MSY} estimates are shown in Table 7.

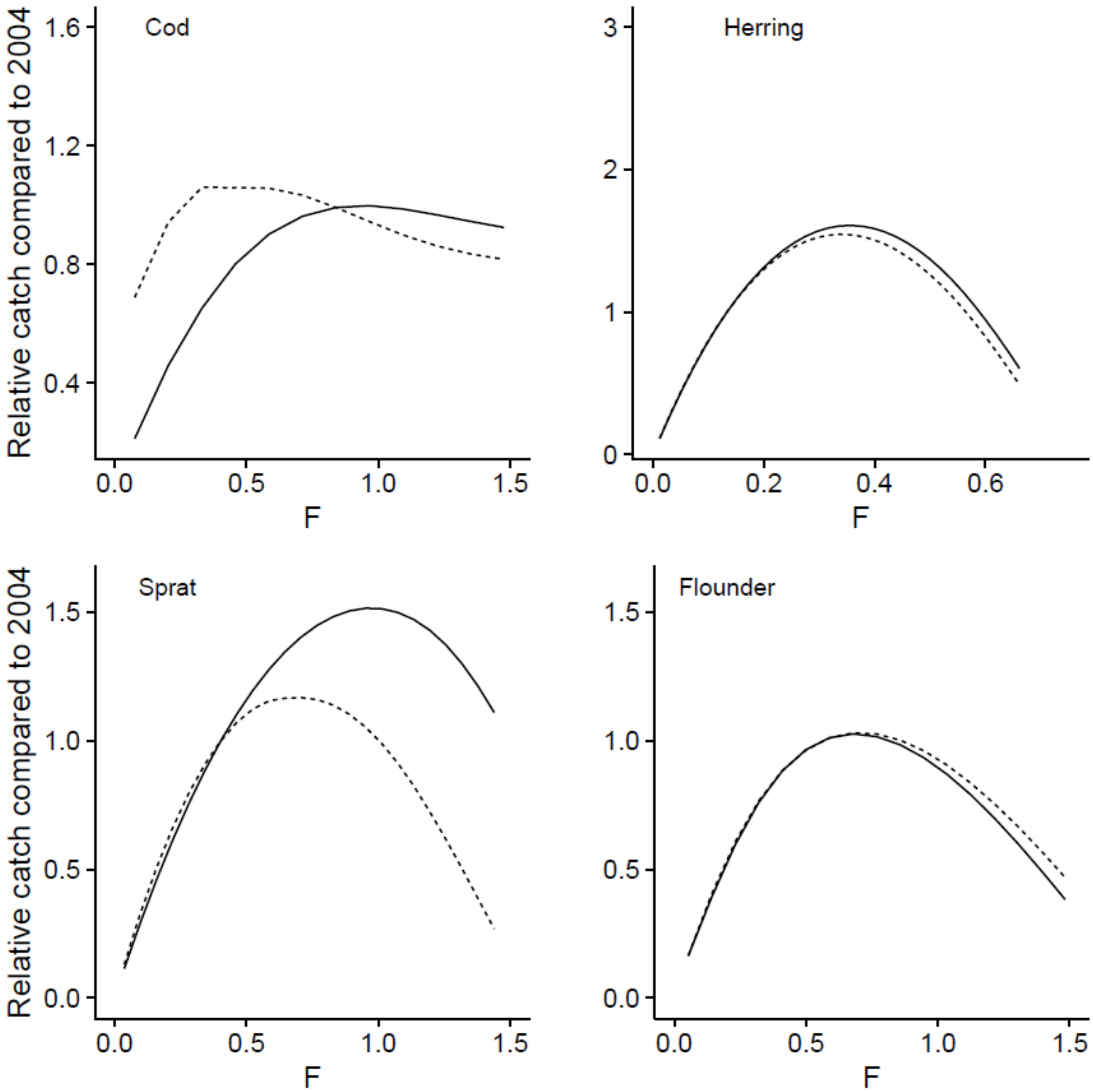


Figure 15. Relative catch as a function of F from Ecosim equilibrium calculations. Dashed: 'stationary', solid: 'full compensation'.

2.3.5 Stability tests

Successful stability testing meant that when not applying any forcing functions and keeping fishing at Ecopath level, all modelled biomasses were stable, and when stopping fishing for a short period only, biomasses returned to stability after continuing fishing again.

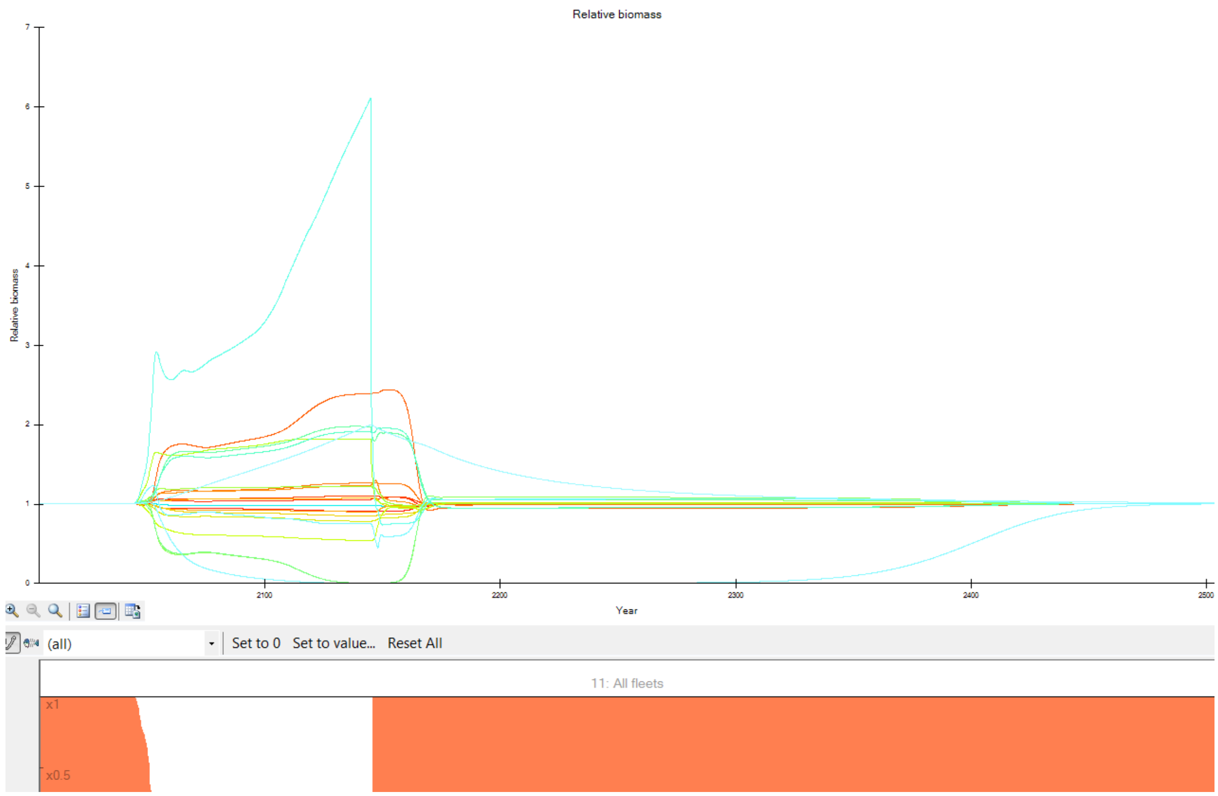


Figure 16. Testing the effects of stopping fishing for a short period, then returning to Ecopath levels. The image shows the Ecosim user interface. The orange field on the bottom indicates the level of fishing effort (multiplier on Ecopath F of each species). Colored lines show relative biomasses of different modelled groups.

We applied also another test, when we switched off fishing completely to test if the system reached an alternative equilibrium. Switching off fishing changed the system to a cod-dominated state. The projected increase in cod biomass is comparable to peak cod biomass levels observed during the 1980s.

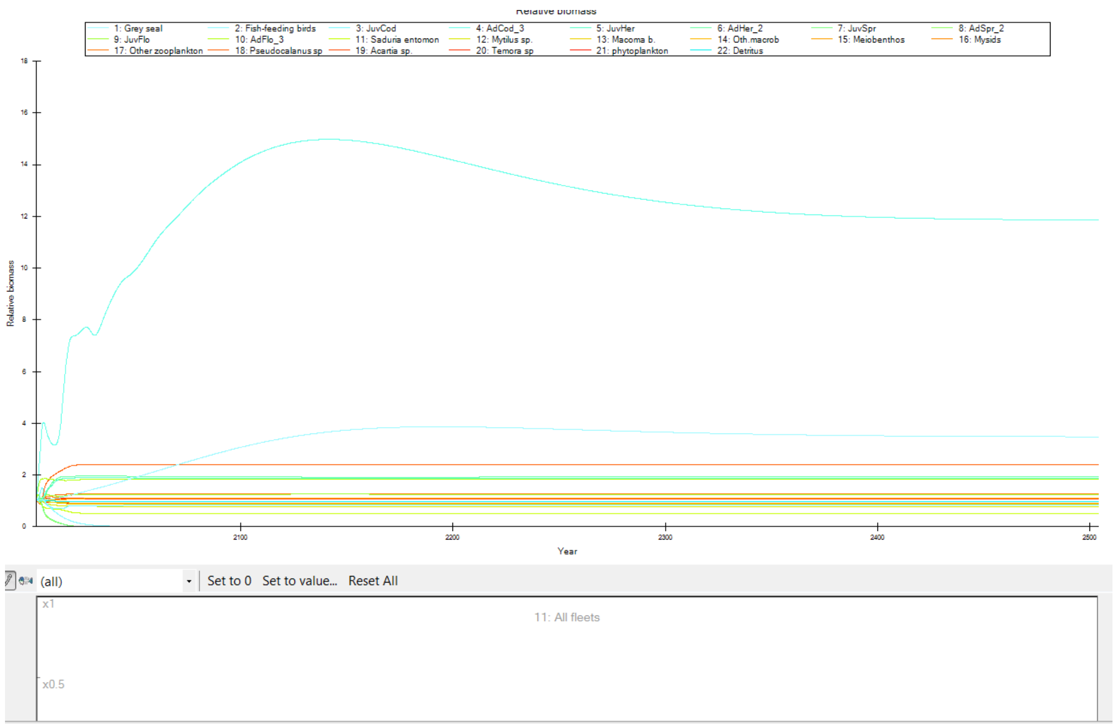


Figure 17. Testing the effects of no fishing. The image shows the Ecosim user interface. Colored lines show relative biomasses of different modelled groups. Light blue line on the top represents adult cod relative biomass.

A general comment needs to be added to the tests applied in Sections 2.3.4 and 2.3.5. Many Ecosim parameters (for example, consumption rates, relative preferences for prey, or prey suitability called 'electivity') are derived from Ecopath and are either fixed (electivities) or can only deviate from Ecopath values to a certain extent (consumption rates). Thus, Ecosim models are best suited to investigate how the 'balanced' system as described by Ecopath changes as a result of a moderate increase or decrease in certain parameters. Predictions of most modelling approaches become more unreliable when conditions or parameters are getting far away from original system conditions and this is also true in the case of EwE models (Plagányi and Butterworth, 2004). Thus, in our opinion, testing the effects of parameters in ranges widely different from Ecopath values, as we have done here with fishing mortalities, only gives an indication of model sensitivity and possible trends, but outcomes can only be interpreted or used as quantitative predictions with great caution.

3. Key run specification and setup

Table 6. Definition of the model setup required to reproduce the Key Run.

ECOPATH VERSION	VERSION 6.5.14040.0 (6.5 OFFICIAL RELEASE)
Database name	Baltic2004_keyrun65.eweaccdb
Ecopath Model name	2000sFishModel

Time-series file name	20161003_keyrun.csv
Ecosim scenario name	fit_WGIAB
Fishing time-series	Yes
PP force	Yes
Consumer forcing	Yes
Sums of squares	858.6
number of time-series fitted to	29 (no forced biomass time-series)

4. Key Run Outputs

4.1 Model fits to data

The fit of model predictions to data are shown on Figure 18, Figure 19 and Figure 20.

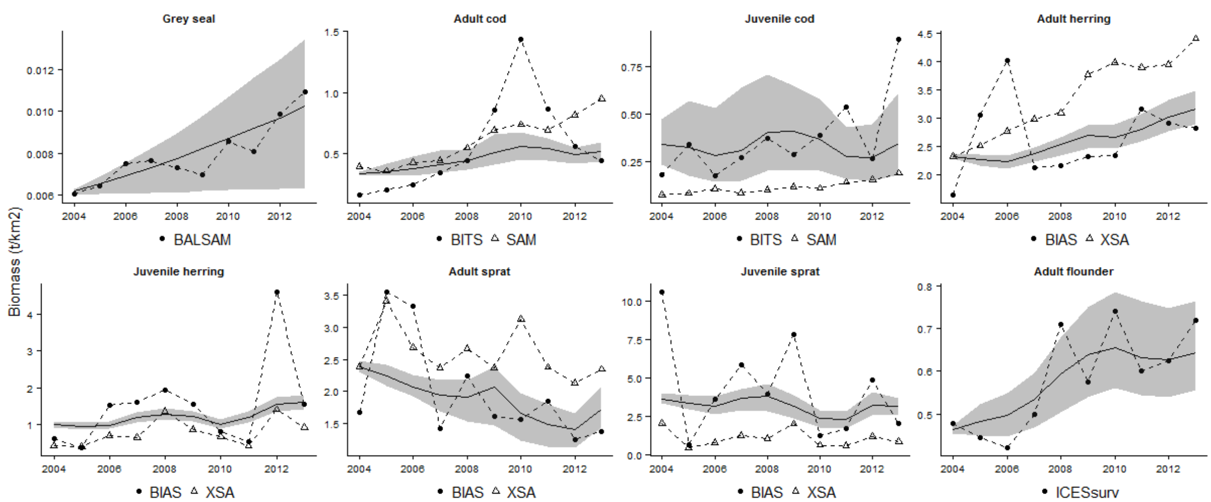


Figure 18. Ecosim predicted biomasses of higher trophic level species (solid lines) fit to time-series of surveys (black circles, dashed lines) after final calibration of vulnerabilities. Fish survey indices of abundance (BITS, BIAS, ICESsurv) are rescaled for visualization. Grey areas indicate 90% confidence intervals based on 100 trials Monte Carlo simulations using default settings, varying Ecopath input biomasses, P/B and Q/B parameters and ecotrophic efficiency of mysids.

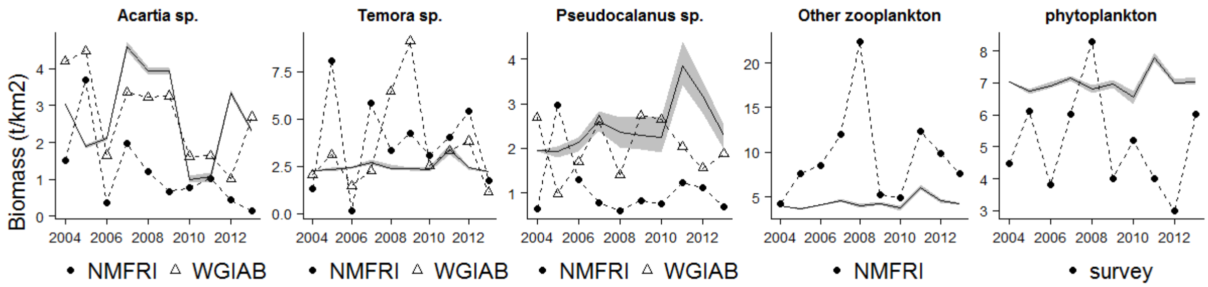


Figure 19. Ecosim predicted biomasses of lower trophic level species (solid lines) fit to time-series of surveys (black circles, dashed lines) after final calibration of vulnerabilities. Grey areas indicate 90% confidence intervals based on Monte Carlo simulations varying Ecopath input biomasses, P/B and Q/B parameters and ecotrophic efficiency of mysids.

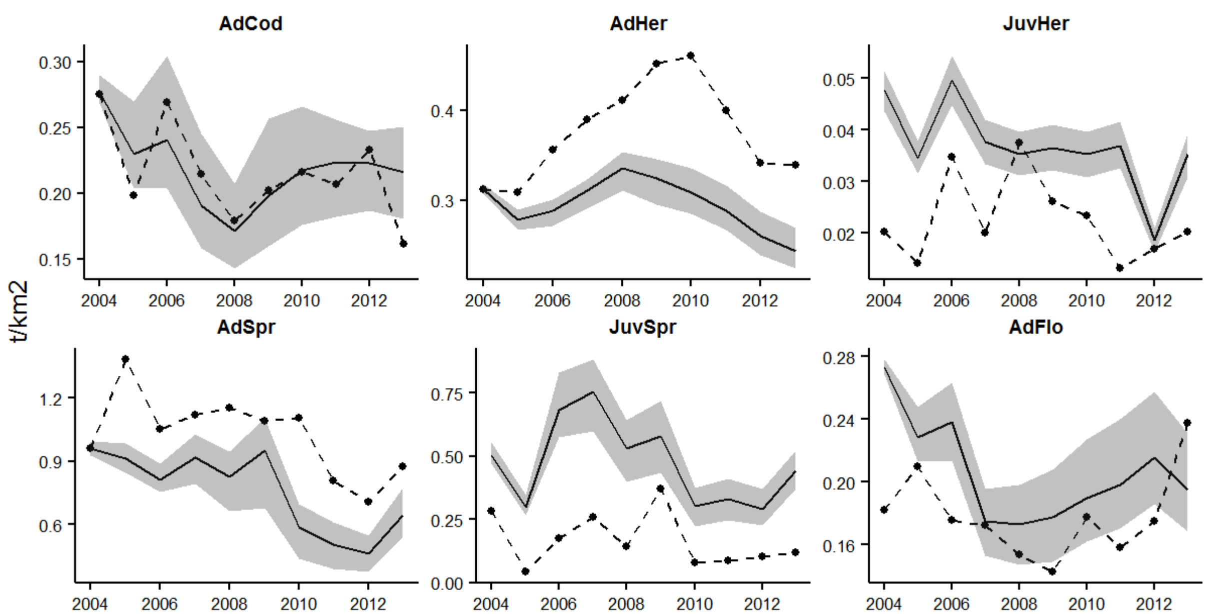


Figure 20. Ecosim predicted catches (solid lines) fit to time-series of catches/landings for flounder (black circles, dashed lines) after final calibration of vulnerabilities. Grey areas indicate 90% confidence intervals based on Monte Carlo simulations varying Ecopath input biomasses, P/B and Q/B parameters and ecotrophic efficiency of mysids.

4.2 Cod diet

Relative percentage of various prey (by weight) in adult (Figure 21) and juvenile (Figure 22) cod diet in stomach data and modelled by Ecosim. The modelled time period is too short to see any trends, except of a decline of *Saduria entomon* in the diet, which is also seen in Ecosim predictions. As described in Section 1.2, diet data was used to parameterize Ecopath, but we increased the amount of sprat in the diet and compensated that by a decrease in *Saduria entomon*, herring and cannibalism.

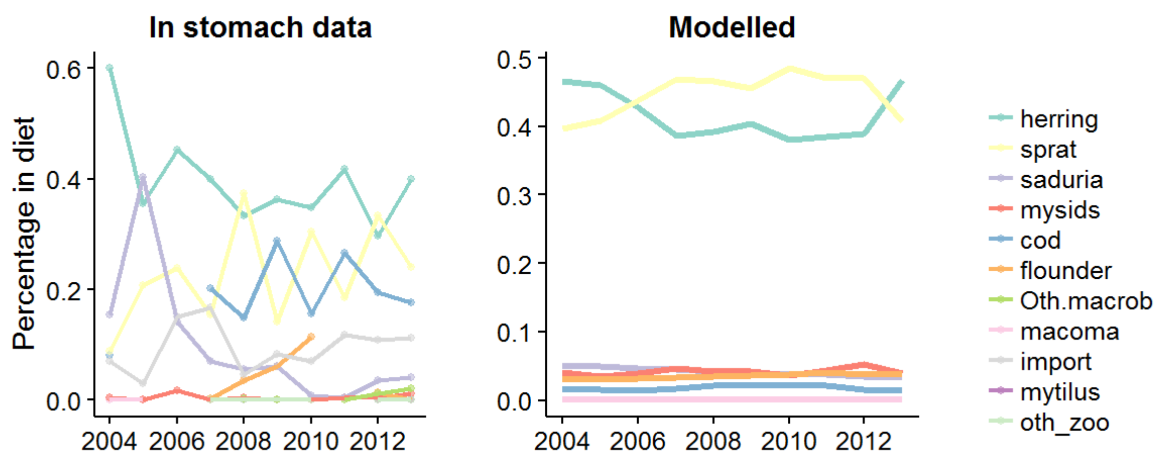


Figure 21. Adult cod diet.

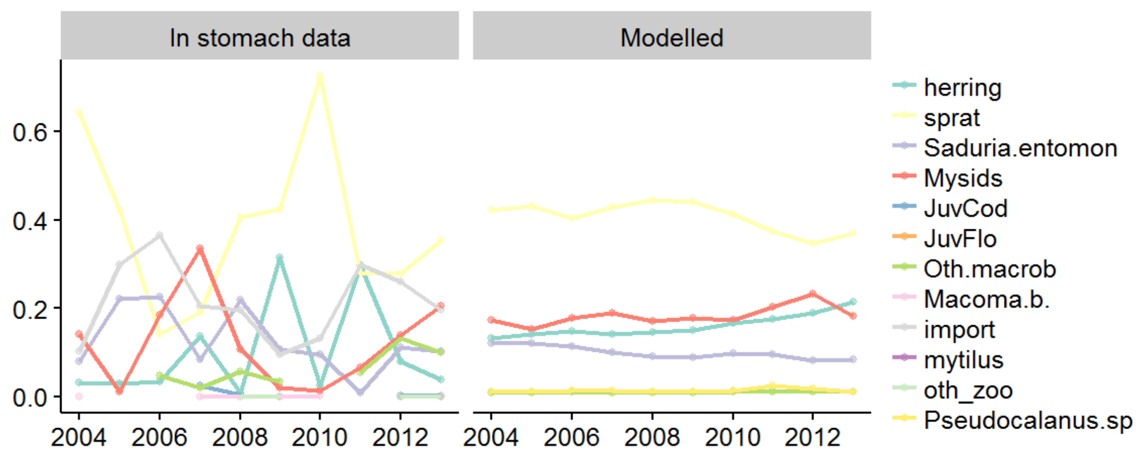


Figure 22. Juvenile cod diet.

4.3 Mortality

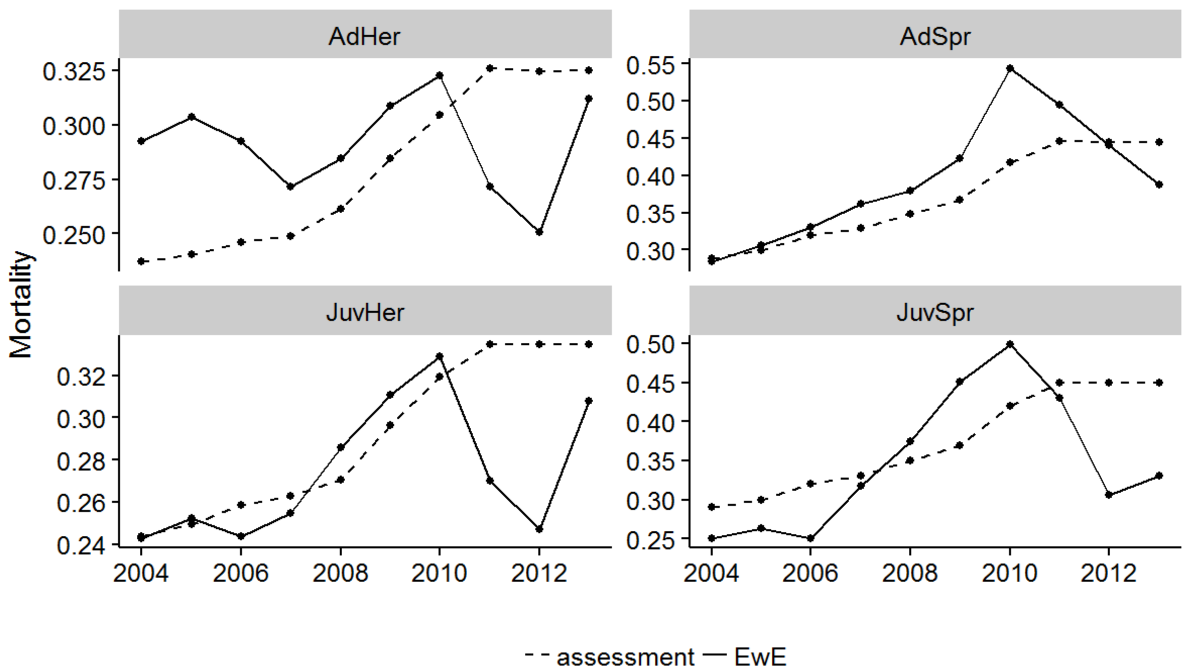


Figure 23. Biomass-weighted average of natural mortality (NATMOR, dashed line) used as XSA input for Age 1 (juveniles) and Ages 2+ (adults) comparison to mortality rate due to predation by cod in EwE (solid). NATMOR is not fitted in the calibration in Ecosim, however, diet proportion of sprat and herring in Ecopath is adjusted considering NATMOR, due to unreliabilities in stomach data (see Section 1.2).

4.4 Equilibrium estimates of F_{MSY}

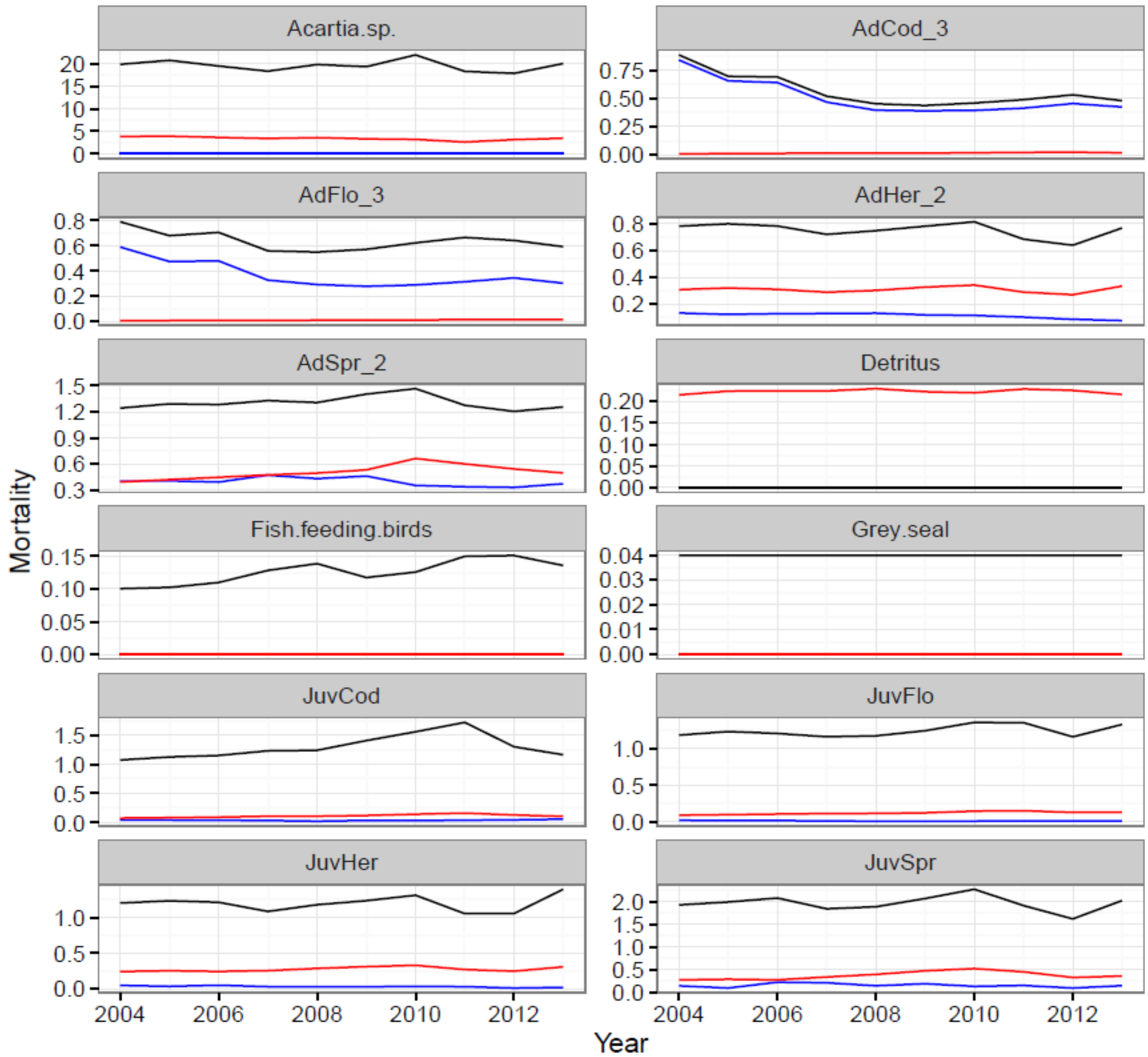
Equilibrium simulations in EwE can be used to simply investigate the relationship between a large range of fishing mortality rates and the corresponding catches, and thereby determine the F which corresponds to the largest catch (“F_{MSY}”). For more details see Section 2.3.4.

Table 7. F’s resulting in highest catch at equilibrium (‘F_{MSY}’) predicted by EwE (first two columns) compared to values reported in *ICES (2016b)*, last two columns.

Group	F_{MSY} stationary	F_{MSY} fullcomp	F_{MSY}	Other
Juv. cod	0.25	0.25		
Adult cod	0.33	0.97	At present not defined, last accepted value (ICES, 2013) 0.46.	At present not defined, last accepted value (ICES, 2013): Multi-species F_{MSY} (SMS) 0.55
Juv. herring	0.1	0.1		
Adult herring	0.34	0.36	0.22	MSY $F_{lower-upper}$ (AR): 0.16-0.28

Juv. sprat	0.38	0.38		
Adult sprat	0.71	0.95	0.26	MSY $F_{lower-upper}$ (AR): 0.19-0.27
Juv. flounder	0.1	0.1		
Adult flounder	0.68	0.68		

4.5 Mortality rates time-series – predation and fishery (partial F's)



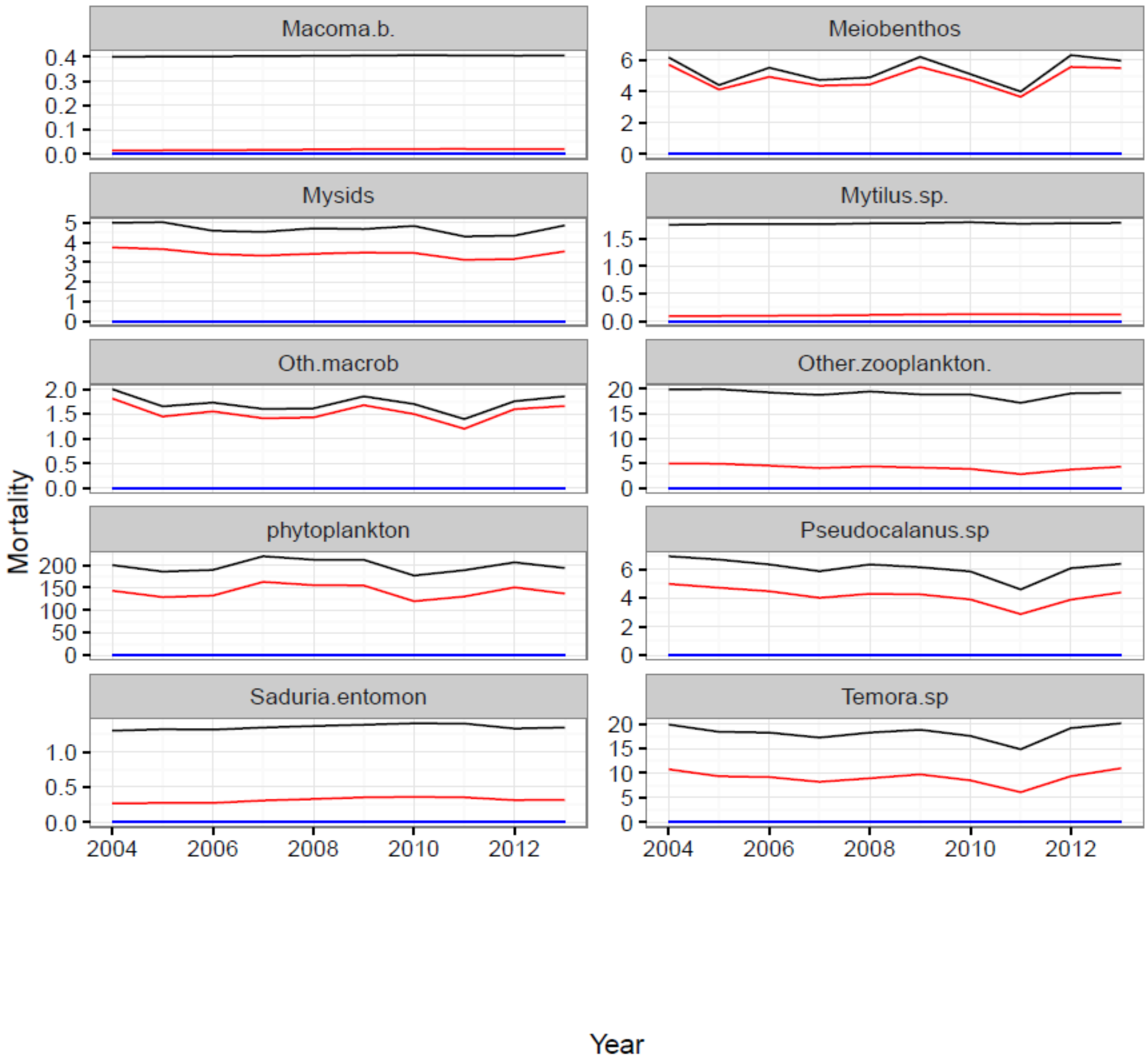


Figure 24. Changes in fishing (blue), predation (red) and total mortality (black).

4.5 Ecosystem indicator trends

All output files of ecosystem metrics describing the state of the Baltic Sea ecosystem in 2004, and changes 2004-2013 in system and community level indicators are given in 2 files

1. Baltic 2004_Key Run_Ecopath Output.xlsx, Ecosystem indicators

2. Baltic 2004_Key Run_Ecosim Output.xlsx, Ecosystem indicators

Changes in indicators are shown in Figure 25. Referring to the figure panels, these include:

(a) Trends in total system biomass and biomass of demersal fish, pelagic fish and benthos.

(b) Community indices – demersal/pelagic fish and fish/benthos

(c) Fish biomass and catch

(d) Total catch/biomass- as a measure of overall fishing pressure

(e) Biomass-weighted mean trophic level of all fish and mean trophic level of catch.

(f) Mean trophic level of all consumers.

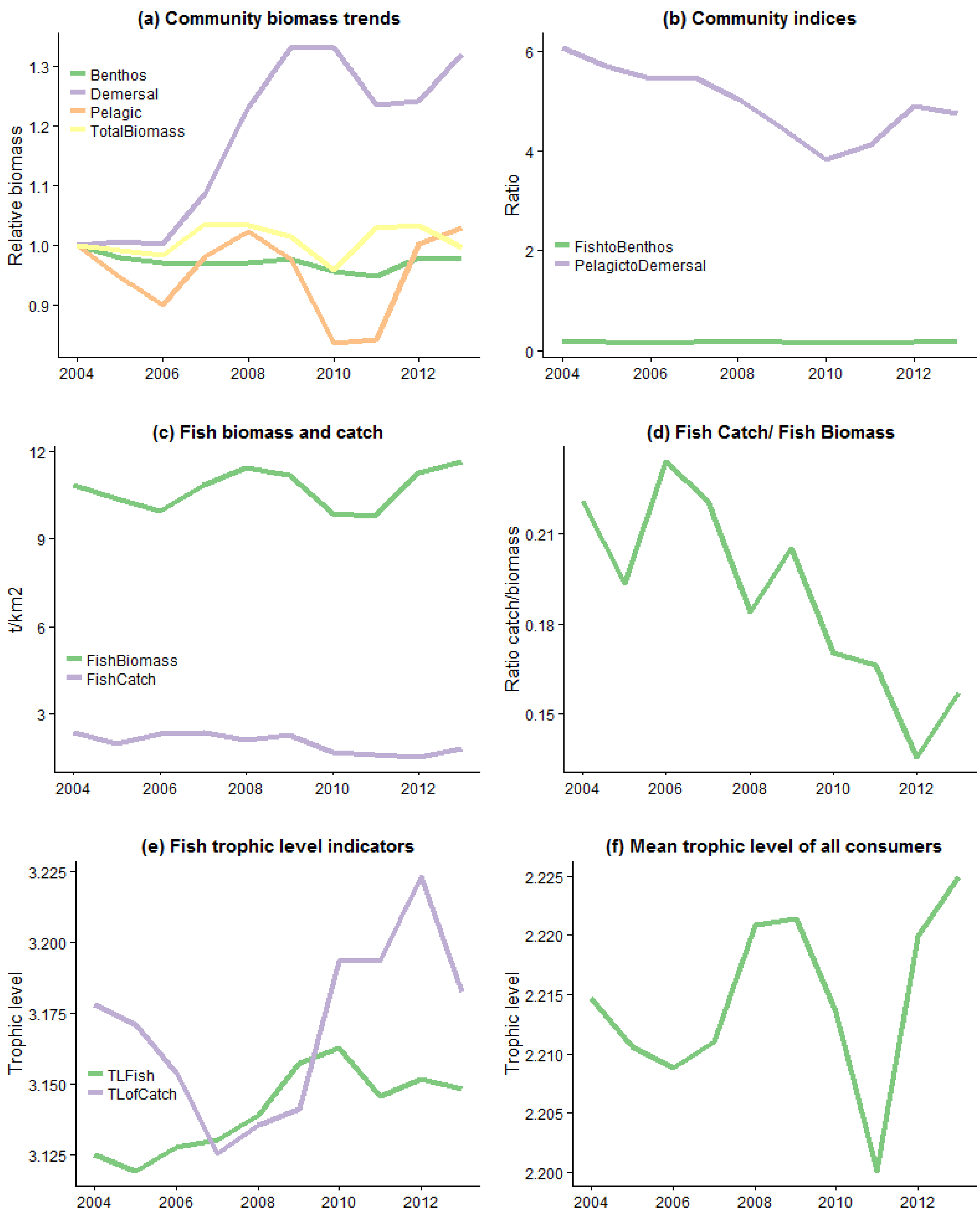


Figure 25. Ecosystem indicators derived from the model key run.

5. References

Aarnio, K., Bonsdorff, E., Rosenback, N., 1996. Food and feeding habits of juvenile flounder platichthys flesus (l.), and turbot scophthalmus maximus l. in the ~land archipelago, northern baltic sea 36, 311–320.

Borg, J.P.G., Westerbom, M., Lehtonen, H., 2014. Sex-specific distribution and diet of Platichthys flesus at the end of spawning in the northern Baltic Sea 937–951. doi:10.1111/jfb.12326

Casini, M., Cardinale, M., 2004. Feeding preferences of herring (Clupea harengus) and sprat (Sprattus sprattus) in the southern Baltic Sea 1277. doi:10.1016/j.icesjms.2003.12.011

Casini, M., Hjelm, J., Molinero, J.-C., Lövgren, J., Cardinale, M., Bartolino, V., Belgrano, A., Kornilovs, G., 2009. Trophic cascades promote threshold-like shifts in pelagic marine ecosystems. Proc. Natl. Acad. Sci. U. S. A. 106, 197–202. doi:10.1073/pnas.0806649105

Casini, M., Lövgren, J., Hjelm, J., Cardinale, M., Molinero, J.-C., Kornilovs, G., 2008. Multi-level trophic cascades in a heavily exploited open marine ecosystem. Proc. R. Soc. London B Biol. Sci. 275, 1793–1801. doi:10.1098/rspb.2007.1752

Christensen, V., Walters, C.J., Pauly, D., Forrest, R., 2008. Ecopath with Ecosim Version 6. User Guide. Fisheries Centre, University of British Columbia, Vancouver, Canada.

Durinck, J., Skov, H., Jensen, F.P., Pihl, S., 1994. Important marine areas for wintering birds in the Baltic Sea.

Enstipp, M.R., Daunt, F., Wanless, S., Humphreys, E.M., Hamer, K.C., Benvenuti, S., Grémillet, D., 2006. Foraging energetics of North Sea birds confronted with fluctuating prey availability, in: Boyd, I.L. *et al.* (Ed.), Top Predators in Marine Ecosystems: Their Role in Monitoring and Management, Conservation Biology 12. pp. 191–210.

Florin, A.B., Lavados, G., 2010. Feeding habits of juvenile flatfish in relation to habitat characteristics in the Baltic Sea. Estuar. Coast. Shelf Sci. 86, 607–612. doi:10.1016/j.ecss.2009.11.028

Gårdmark, A., Östman, Ö., Nielsen, A., Lundström, K., Karlsson, O., Pönni, J., Aho, T., 2012. Does predation by grey seals (Halichoerus grypus) affect Bothnian Sea herring stock estimates? ICES J. Mar. Sci. J. du Cons. doi:10.1093/icesjms/fss099

Gustafsson, B.G., 2003. A time-dependent coupled-basin model of the Baltic Sea. Göteborg University, Göteborg.

Haahtela, I., 1990. What do Baltic studies tell us about the isopod Saduria entomon (L.)? Ann. Zool. Fennici 27, 269–278.

Härkönen, T., Galatius, A., Bräger, S., Karlsson, O., Ahola, M., 2013. Population growth rate , abundance and distribution of marine mammals. HELCOM Core Indic. Report. Online 1–34.

Harris, M.P., Wanless, S., Rothery, P., Swann, R.L., Jardine, D.C., 2000. Survival of adult common guillemots Uria aalge at three Scottish colonies. Bird Study 47, 1–7. doi:10.1080/00063650009461154

Harvey, C.J., Cox, S.P., Essington, T.E., Hansson, S., Kitchell, J.F., 2003. An ecosystem model of food web and fisheries interactions in the Baltic Sea 3139, 939–950. doi:10.1016/S1054

HELCOM, 2012. Development of a set of core indicators: Interim report of the HELCOM CORESET project. PART B: Descriptions of the indicators. Balt. Sea Environ. Proc. 129.

Herrmann, C., Rintala, J., Lehikoinen, A., Petersen, I.K., Hario, M., Kadin, N., Korpinen, S., 2013. Abundance of waterbirds in the breeding season. HELCOM Core Indicator Report.

Huwer, B., Neuenfeldt, S., Rindorf, A., Andreasen, H., Levinsky, S.-E., Storr-Paulsen, M., Dalmann, R.S., Haslund, O.H., Horbowy, J., Pachur, M., Pawlak, J., Ustups, D., Kruze, E., Sics, I., Uzars, D., Velasco, A., Kempf, A., Eberle, S., Floeter, J., Temming, A., van Hal, R., de Boois, I., Pennock, I., Hoek, R., Pinnegar, J., Hunter, E., Plirú, A., Casini, M., Belgrano, A., 2014. Study on stomach content of fish to support the assessment of good environmental status of marine food webs and the prediction of MSY after stock restoration.

ICES, 2016a. Report of the Workshop on Spatial Analyses for the Baltic Sea (WKSPATIAL), 3-6 November 2015, Rome, Italy. ICES C. 2015/SSGIEA 13, 37.

ICES, 2016b. Report of the Baltic Fisheries Assessment Working Group (WGBFAS), 12- 19 April 2016, ICES HQ, Copenhagen, Denmark. ICES CM 2016/ACOM:11.

ICES, 2015a. Report of the Baltic Fisheries Assessment Working Group (WGBFAS), 14- 21 April 2015, ICES HQ, Copenhagen, Denmark. ICES C. 2015/ACOM 10, 806.

ICES, 2015b. Report of the ICES/HELCOM Working Group on Integrated Assessments of the Baltic Sea (WGIAB), 9-13 March 2015, Cádiz, Spain. ICES CM 2015/SSGIEA:08.

ICES, 2013. Report of the Baltic Fisheries Assessment Working Group (WGBFAS), 10 - 17 April 2013, ICES Headquarters. ICES C. 2013/ACOM 10, 747.

ICES, 2012. Report of the Second ICES/HELCOM Workshop on Flatfish in the Baltic Sea, 19 - 23 March 2012, ICES Headquarters, Copenhagen. ICES C. 2012/ACOM 33, 135.

ICES, 2011. ICES WGSAM REPORT 2011 Report of the Working Group on Multispecies Assessment Methods (WGSAM) International Council for the Exploration of the Sea.

ICES, 2008. Report of the Working Group on Integrated Assessment of the Baltic Sea (WGIAB), 25–29 March 2008, Öregrund, Sweden. CM 2008/BCC:04.

Lavers, J., Hipfner, M., Chapdelaine, G., Hipfner, M.J., 2009. Razorbill (Alca torda) [WWW Document]. Birds North Am. Online (A. Poole, Ed.), Ithaca Cornell Lab Ornithol. doi:10.2173/bna.635

Lilliendahl, K., Solmundsson, J., 1997. An estimate of summer food consumption of six seabird species in Iceland. ICES J. Mar. Sci. – ICES J. Mar. Sci. 54, 624–630. doi:10.1006/jmsc.1997.0240

Link, J.S., 2010. Adding rigor to ecological network models by evaluating a set of pre-balance diagnostics: A plea for PREBAL. Ecol. Modell. 221, 1580–1591. doi:10.1016/j.ecolmodel.2010.03.012

Mehlum, F., Gabrielsen, G.W., 1993. Energy expenditure by black guillemots (Cepphus grylle) during chick-rearing. Colon. Waterbirds 16, 45–52. doi:10.2307/1521555

Möllmann, C., Kornilovs, G., Fetter, M., Köster, F.W., 2004. Feeding ecology of central Baltic Sea herring and sprat. J. Fish Biol. 65, 1563–1581. doi:10.1111/j.1095-8649.2004.00566.x

Möllmann, C., Müller-Karulis, B., Kornilovs, G., St John, M. a., 2008. Effects of climate and overfishing on zooplankton dynamics and ecosystem structure: Regime shifts, trophic cascade, and feedback loops in a simple ecosystem. ICES J. Mar. Sci. 65, 302–310. doi:10.1093/icesjms/fsm197

Niiranen, S., Yletyinen, J., Tomczak, M.T., 2013. Combined effects of global climate change and regional ecosystem drivers on an exploited marine food web 1–16. doi:10.1111/gcb.12309

Nissling, A., Jacobsson, M., Hallberg, N., 2007. Feeding ecology of juvenile turbot Scophthalmus maximus and flounder Pleuronectes flesus at Gotland, Central Baltic Sea. J. Fish Biol. 70, 1877–1897. doi:10.1111/j.1095-8649.2007.01463.x

Österblom, H., Casini, M., Olsson, O., Bignert, A., 2006. Fish , seabirds and trophic cascades in the Baltic Sea 323, 233–238.

Österblom, H., Fransson, T., Olsson, O., 2002. Bycatches of common guillemot (Uria aalge) in the Baltic Sea gillnet fishery. Biol. Conserv. 105, 309–319. doi:10.1016/S0006-3207(01)00211-7

Österblom, H., Hansson, S., Larsson, U., Hjerne, O., Wulff, F., Elmgren, R., Folke, C., 2007. Human-induced trophic cascades and ecological regime shifts in the baltic sea. Ecosystems 10, 877–889. doi:10.1007/s10021-007-9069-0

Plagányi, É.E., Butterworth, D.S., 2004. A critical look at the potential of Ecopath with ecosim to assist in practical fisheries management. African J. Mar. Sci. 26, 261–287. doi:10.2989/18142320409504061

R Core Team, 2016. R: A Language and Environment for Statistical Computing.

Scott, E., Serpetti, N., Steenbeek, J., Heymans, J.J., 2016. A Stepwise Fitting Procedure for automated fitting of Ecopath with Ecosim models. SoftwareX. doi:http://dx.doi.org/10.1016/j.softx.2016.02.002

Timmermann, K., Norkko, J., Janas, U., Norkko, A., Gustafsson, B.G., Bonsdorff, E., 2012. Modelling macrofaunal biomass in relation to hypoxia and nutrient loading. J. Mar. Syst. 105–108, 60–69. doi:10.1016/j.jmarsys.2012.06.001

Tomczak, M.T., Niiranen, S., Hjerne, O., Blenckner, T., 2012. Ecosystem flow dynamics in the Baltic Proper-Using a multi-trophic dataset as a basis for food-web modelling. Ecol. Modell. 230, 123–147. doi:10.1016/j.ecolmodel.2011.12.014

Ustups, D., Uzars, D., Mueller-Karulis, B., 2007. Structure and feeding ecology of the fish community in the surf zone of the eastern Baltic Latvian coast. Proc. Latv. Acad. Sci. Sect. B Nat. Exact Appl. Sci. 61, 96–106.

Walters, C., Christensen, V., Walters, W., Rose, K., 2010. REPRESENTATION OF MULTISTANZA LIFE HISTORIES IN ECOSPACE MODELS FOR SPATIAL ORGANIZATION 86, 439–459.

Witek, Z., 1995. Biological Production and its Utilisation within a Marine Ecosystem in the Western Gdansk Basin. Sea Fisheries Institute, Gdynia, Poland.